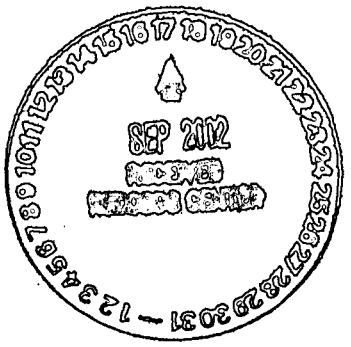
United States Environmental Protection Agency | Office of Solid Waste Washington, DC 20460 | EPA 530-R-99-019a November 1999

EPA

Revised Risk Assessment for the Air Characteristic Study Volume I Overview



Best Available Copy
DOCUMENT CLASSIFICATION REVIEW WAIVER PER CLASSIFICATION OFFICE
ADMIN RECORD
SW-A-004560

1/127

EPA 530-R-99-19a
November 1999

Revised Risk Assessment for the Air Characteristic Study Volume I Overview

Office of Solid Waste
U.S. Environmental Protection Agency
Washington, DC 20460

Printed on Recycled Paper

Table of Contents

Section			Page
Executive Summary			ES-1
1.0	Introduction		1-1
	1.1	Purpose and Requirements of the Air Characteristic Study	1-1
	1.2	Overview of Risk Assessment	1-2
	1.3	Organization of Report	1-6
	1.4	Companion Documents	1-6
2.0	Revisions to the Risk Assessment Framework		2-1
	2.1	Source Characterization	2-1
		2.1.1 Landfills	2-1
		2.1.2 Land Application Units	2-2
		2.1.3 Wastepiles	2-3
		2.1.4 Tanks	2-3
	2.2	Emissions Modeling	2-4
	2.3	Air Dispersion Modeling	2-7
	2.4	Human Health Benchmarks	2-10
	2.5	Exposure and Risk Modeling	2-12
3.0	Summary of Risk Assessment Modeling Approach and Data Sources		3-1
	3.1	Overview of Modeling Approach	3-1
	3.2	Conducting the Analysis	3-5
		3.2.1 Data Sources	3-10
		3.2.2 Emissions Modeling	3-11
		3.2.3 Dispersion Modeling	3-20
		3.2.4 Exposure Modeling/Risk Estimation	3-23
	3.3	Analysis of Variability and Uncertainty	3-33
		3.3.1 Variability	3-33
		3.3.2 Uncertainty	3-33
4.0	Summary of Risk Assessment Results		4-1
	4.1	Overview of Results	4-1
	4.2	Effect of Chemical Properties and Health Benchmarks	4-5
	4.3	Effect of WMU Type	4-33
	4.4	Effect of Exposure Factors	4-34
	4.5	Subchronic and Acute Results	4-34
5.0	Characterization of Significant Findings		5-1
	5.1	Introduction	5-1
	5.2	Gaps in Constituent Coverage	5-2
		5.2.1 RCRA Gaps in Constituent Coverage	5-2
		5.2.2 Clean Air Act Gaps in Constituent Coverage	5-3

3

Table of Contents (continued)

Section		Page
5.2.3	Summary of Gaps in Regulation	5-4
5.3	Air Emission Controls Under RCRA	5-4
5.3.1	Air Emission Controls for Tanks Under RCRA	5-4
5.3.2	Air Emission Controls for Land Units Under RCRA	5-5
6.0	References	6-1
Appendix A -	Comparison with LDR Universal Treatment Standards and Toxicity Characteristic	A-1

4

List of Figures

Figure		Page
2-1	Cumulative tank distributions used in the current study	2-5
3-1	Conceptual diagram of a waste site	3-2
3-2	Model framework	3-6
3-3	Combination of results for individual WMUs into a distribution across all WMUs	3-9
3-4	Meteorological station regions	3-12
4-1	Histogram of most protective 90/90 C_w for aerated treatment tanks	4-2
4-2	Histogram of most protective 90/90 C_w for nonaerated treatment tanks	4-2
4-3	Histogram of most protective 90/90 C_w for storage tanks	4-3
4-4	Histogram of most protective 90/90 C_w for landfills	4-3
4-5	Histogram of most protective 90/90 C_w for LAUs	4-4
4-5	Histogram of most protective 90/90 C_w for wastepiles	4-4

5

List of Tables

Table		Page
1-1	Constituents Modeled for All WMUs	1-3
1-2	Constituents Modeled for Tanks Only	1-5
2-1	Percentiles of New Tank Distributions Associated with Two Model Tanks	2-5
2-2	Chemical-specific Effects of Biodegradation Rate Correction	2-8
2-3	Summary of Issues and Changes for Inhalation Benchmarks Used in the Air Characteristic Study	2-11
3-1	CHEMDAT8 Land-Based Unit Model Input Requirements	3-14
3-2	CHEMDAT8 Tank Model Input Requirements	3-15
3-3	Inhalation Health Benchmarks Used in the Air Characteristic Analysis	3-26
3-4	Summary of Variability and Uncertainty in the Study	3-34
4-1	Chronic 90/90 C_w at 25 m to 1,000 m, Risk = 10^{-5}/HQ = 1 for Landfills (mg/kg)	4-6
4-2	Chronic 90/90 C_w at 25 m to 1,000 m, Risk = 10^{-5}/HQ = 1 for Land Application Units (mg/kg)	4-10
4-3	Chronic 90/90 C_w at 25 m to 1,000 m, Risk = 10^{-5}/HQ = 1 for Wastepiles (mg/kg)	4-14
4-4	Chronic 90/90 C_w at 25 m to 1,000 m, Risk = 10^{-5}/HQ = 1 for Aerated Treatment Tanks (mg/L)	4-18
4-5	Chronic 90/90 C_w at 25 m to 1,000 m, Risk = 10^{-5}/HQ = 1 for Nonaerated Treatment Tanks (mg/L)	4-23
4-6	Chronic 90/90 C_w at 25 m, Risk + 10^{-5}/HQ = 1 for Storage Tanks (mg/L)	4-27
4-7	Physical-Chemical Properties and Health Benchmarks	4-31
4-8	Most Limiting WMU Type	4-34
4-9	Acute and Subchronic 100/90 C_w at 25 to 75 m for HQ = 1 for Land Application Units (mg/kg)	4-36
4-10	Acute and Subchronic 100/90 C_w at 25 to 75 m for HQ = 1 for Wastepiles (mg/kg)	4-40
4-11	Comparison of C_w for Chronic, Subchronic, and Acute Averaging Times for HQ = 1 for Land Application Units (mg/kg)	4-44
4-12	Comparison of C_w for Chronic, Subchronic, and Acute Averaging Times for HQ = 1 for Wastepiles (mg/kg)	4-47
5-1	Regulatory, Occurrence, and Risk Comparison for Tanks—Constituents with Waste Concentrations Less than 100 ppm	5-6
5-2	Regulatory, Occurrence, and Risk Comparison for Land-Based Units—Constituents with Waste Concentrations Less than 100 ppm	5-9

6

Executive Summary

The U.S. Environmental Protection Agency (EPA), Office of Solid Waste, has analyzed the potential direct inhalation risks that may result from unregulated emissions from certain waste management units. This document (Volume I) presents an overview of the revised risk assessment for that analysis, also referred to as the Revised Risk Assessment for the Air Characteristic Study. Volume II is the Technical Background Document and Volume III (on CD-ROM) presents results.

The Air Characteristic Study

This report and the 1998 Air Characteristic Study are among the initial steps for the EPA in fulfilling a long standing goal to review the adequacy and appropriateness of the hazardous waste characteristics.

The first step for EPA in achieving this goal was the Hazardous Waste Characteristic Scoping Study (November 1996), in which the Agency investigated potential gaps in the characteristics. The Scoping Study identified direct inhalation risks from emissions of waste management units as one potential gap in the Resource Conservation and Recovery Act (RCRA) hazardous waste characteristics. The Agency then completed the Air Characteristic Study (May 1998) as the next step in the process. The Air Characteristic Study examined the potential direct inhalation risks due to emissions from certain waste management units. In accordance with Agency policy, the technical work performed for the 1998 Air Characteristic Study was peer-reviewed. This report contains the revised Air Characteristic Risk Assessment based on peer-review and public comments.

Revised Risk Analysis

This study is a national analysis to evaluate the possible need for an air characteristic. As such, this study was designed to highlight areas that may require a more detailed review before any formalized regulatory development work is initiated.

The overall goal of the risk analysis is to estimate waste concentrations that could be present in certain waste management units (WMUs) and still be protective of human health. Concentrations at specified risk levels were estimated at six different distances for a subset of constituents that could be present in wastepiles, landfills, land application units, storage tanks, and aerated and nonaerated treatment tanks. The analysis is based on modeling the emissions from a waste management unit, transport through the ambient environment, and exposure to a receptor to backcalculate to a threshold concentration in waste below which the risk to human health would fall below a pre-established threshold. To accomplish this, we characterized waste sources, applied peer-reviewed and commonly used emissions and dispersion models, and established a Monte Carlo analysis to capture variabilities in receptor characteristics, such as exposure parameters and location around a facility. Chronic exposures were evaluated for 104 of

7

the 105 constituents,* and acute and subchronic exposures were considered for 35 and 64 constituents, respectively. In addition, protective concentrations in waste were estimated for five receptor categories: an adult resident, a child resident with exposure starting between 0 and 3 years old, a child resident with exposure starting between 4 and 10 years old, a child resident with exposure starting between 11 and 18 years old, and an off-site worker.

Distances of 25, 50, 75, 150, 500, and 1,000 meters were used as the basis for backcalculated risk-based waste concentrations. The resulting waste concentrations were considerably higher for receptors at the 500- and 1,000-m distances. A sensitivity analysis conducted on the dispersion component of this analysis indicated that there is a sharp decline in air concentration after the 150-m distance. The 25-m distance produces the lowest waste concentrations but is also an unlikely exposure scenario. The 50-, 75- and 150-m results were very similar to each other (within a factor of 2 to 3). This report displays results for only the 25-, 150- and 1,000-m distances. Results for the remaining distances are provided in Volume III: *Results*.

Results of the risk analysis indicate that the lowest estimated protective waste concentrations (e.g., highest risk) were for the aerated and nonaerated treatment tanks. Aeration increases the potential for a chemical to be emitted to the air, which results in a higher emission rate per unit area for these tanks relative to the other units. Nonaerated tanks are typically bigger than aerated tanks, resulting in similar total emissions. In general, the estimated protective waste concentrations for treatment tanks were lower than the other units by about an order of magnitude or more. Following aerated and nonaerated treatment tanks, the WMU ranking was storage tanks, land application units, landfills, and wastepiles.

Of the receptors evaluated, the protective waste concentrations for adult residents were lowest (i.e., highest risk), followed by the child residents, from youngest to oldest. The estimated waste concentrations for the offsite worker were about an order of magnitude higher than those for residents. The differences in the results for the resident scenarios can be attributed to the variation in assumed exposure duration. The exposure duration used in the risk modeling was greatest for the adult, followed by the child residents, and finally the off-site worker. For the chronic exposures, it appeared that the most important factor affecting the results was the chemical's toxicity. The chemicals with the lowest protective waste concentrations, and so highest risk, were among the most toxic.

No clear pattern emerged from the chronic, subchronic, and acute results. Subchronic and acute results may be lower or higher than chronic results depending on the chemical, and the difference ranges from negligible up to 2 orders of magnitude in either direction. The most likely reason for this is that the hazard posed by a chemical is likely to vary with exposure duration, i.e., some chemicals have greater hazard at chronic exposures; others at acute and subchronic exposures.

* Note that one chemical of the original 105, 3,4-dimethylphenol, was addressed, but risks could not be quantified because data were insufficient to develop a health benchmark.

6

Integrating the Revised Risk Assessment with the 1998 Analyses

In order to determine the need for an Air Characteristic, the Agency conducted two other analyses in 1998 along with the risk assessment. These analyses on regulatory coverage and constituent occurrence were to ascertain the current management of the 105 constituents. Integrating the results from these two analyses with the risk assessment results would help the Agency identify the nature and extent of gaps in regulatory coverage and the significance of the resulting human health risks.

This step was repeated for this task. The results from the revised risk assessment were combined with the results of the regulatory gaps analysis and the occurrence analysis from the 1998 Air Characteristic Study. This comparison showed that 16 constituents were neither associated with a listing nor on the Toxicity Characteristic (TC) list under RCRA. Two of these constituents had concentrations in tanks less than 100 ppm. In addition, 2 of these 16 constituents were not on the Clean Air Act's hazardous air pollutants (HAPs) list.

Three constituents had estimated protective waste concentrations lower than the TC or TC-derived waste concentration. Two constituents had TC levels that may not be protective of air pathway risks for tanks, and two constituents had waste concentrations more stringent than TC levels for land-based units. The magnitude of the difference between the TC and the estimated air characteristic waste concentrations (C_w) varied according to the waste management unit and the constituent.

Land disposal restrictions (LDRs) and the protective concentrations in waste were compared, and results indicated that the treatment standards are not always below the levels at which there are potential air risks. Two constituents had concentrations in waste for chronic exposures that were below the LDR treatment levels. No constituents had concentrations in waste that were below the LDR treatment levels for acute or subchronic exposures.

Next Steps

Should EPA decide this analysis identifies constituents and waste management units of potential significance as unregulated emissions of possible concern, EPA has a range of options. EPA could decide to further study and potentially address these issues through regulation under the CAA, RCRA, or both. Further analysis would be needed before any new regulatory action could be promulgated.

9

1.0 Introduction

The U.S. Environmental Protection Agency (EPA), Office of Solid Waste (OSW), has analyzed the potential risks to human health posed by the inhalation of vapor (gaseous) and particulate (nongaseous) air emissions from a set of chemicals and metals when managed in certain waste management units (WMUs). An analysis of these risks was initially performed in 1998 as part of the *Air Characteristic Study* (U.S. EPA, 1998a). In accordance with Agency policy, the risk assessment conducted for the 1998 Air Characteristic Study was peer reviewed to ensure that science was used credibly and appropriately in the work performed. Based on comments made by the peer reviewers, EPA has revised the original risk assessment.

This report presents the revised risk assessment in three volumes. This document is Volume I, the Overview. This volume provides a discussion of the changes made from the 1998 Air Characteristic Study, a general overview of the risk assessment, a summary of results of the risk assessment, and the integration of the revised risk assessment results with the May 1998 regulatory gaps and occurrence analyses. A detailed description of the methodologies, data, and supporting analyses used for the risk assessment may be found in Volume II, *Revised Risk Analysis for the Air Characteristic Study: Technical Background Document.* The complete results of the analysis are presented in Volume III, *Revised Risk Analysis for the Air Characteristic Study: Results* (on CD-ROM).

1.1 Purpose and Requirements of the Air Characteristic Study

This report and the 1998 Air Characteristic Study are among the initial steps for EPA in fulfilling a long-standing goal to review the adequacy and appropriateness of the hazardous waste characteristics. The first step in achieving this goal was the Hazardous Waste Characteristic Scoping Study (U.S. EPA, 1996), which the Agency completed November 15, 1996, under a deadline negotiated with the Environmental Defense Fund. This study was conducted to identify potential gaps in the current hazardous waste characteristics, as well as other modifications and updates that are necessary to ensure that the definition of characteristics is complete, up-to-date, and based on state-of-the-art methodologies. Based on the initial bounding analysis of potential risks due to air emissions done as part of the Scoping Study, as well as follow-up analysis on potential gaps in regulatory coverage under the Clean Air Act (CAA) and Subpart CC of the Resource Conservation and Recovery Act (RCRA), OSW identified air emissions from WMUs as one of the areas meriting further analysis.

The Air Characteristic Study addresses this area by examining the potential direct inhalation risks due to emissions from certain WMUs. On May 15, 1998, in accordance with a consent decree, EPA completed the first portion of the study. According to the consent decree with EDF, a second part of the Air Characteristic Study, covering surface impoundments

10

receiving wastewaters that never exhibited a characteristic, will be completed March 26, 2001. The purpose of the 1998 Air Characteristic Study, as outlined by the consent decree, was to investigate gaps in the current hazardous waste characteristics and CAA programs. In addition, resulting potential risks to human health posed by the inhalation of air emissions from wastes managed in certain WMUs were to be investigated.

The 1998 Air Characteristic Study has three components: an evaluation of the coverage and potential regulatory gaps in RCRA Subtitle C and the CAA, a risk analysis of air emissions from WMUs, and an evaluation of the occurrence of these constituents in nonhazardous industrial waste. The risk assessment component has undergone a peer review, and EPA has made a number of changes to the risk assessment based on peer reviewer comments. In addition, other revisions have been made based on public comments and improvements initiated by the Agency. Since the other components of the May 1998 Air Characteristic Study have not been revised, those analyses are not covered in this document. The original results of these analyses are used in this report to present the significant findings from the integration of the revised risk assessment results with the regulatory gaps and occurrence analyses.

1.2 Overview of Risk Assessment

The risk assessment described in this document is a national analysis designed to assess the potential human health risk attributable to inhalation exposures when certain chemicals and metals are managed as waste in certain types of WMUs. The purpose of the analysis is to determine which chemicals and waste management units are of potential national concern purely from a risk perspective; it is not intended to draw conclusions concerning regulatory coverage. This information, combined with preliminary information on regulatory coverage and on the presence of these chemicals in nonhazardous waste, will be useful in determining the need for expanded regulatory coverage. Specifically, the purpose of this study is to provide technical information on the potential risk from WMU emissions to help EPA determine the need to expand regulatory coverage in the future.

The analysis presented in this report addresses specific chemicals that when managed as a waste may pose a risk through direct inhalation exposures. Tables 1-1 and 1-2 list the chemicals and metals included in this analysis. The analysis is structured so that the results of the risk assessment are the concentrations of each constituent that can be present in each type of WMU and still be protective of human health. The protective concentrations in waste were developed for three types of receptors: adult residents, child residents, and workers. Three risk endpoints— chronic (over 1 year), subchronic (1 month), and acute (1 day)— were evaluated.

The protective waste concentrations were estimated by modeling the emissions from a waste management unit, the transport through the ambient environment, and the exposure to a receptor to backcalculate a threshold concentration in a waste below which the risk to human health would fall below a pre-established threshold. The waste management scenario modeled in this analysis is storage, disposal, or treatment of industrial waste streams in RCRA subtitle D WMUs.

11

Table 1-1. Constituents Modeled for All WMUs

Constituent	CAS No.
Acetaldehyde [ethanal]	75-07-0
Acetone [2-propanone]	67-64-1
Acetonitrile [methyl cyanide]	75-05-8
Acrolein	107-02-8
Acrylonitrile	107-13-1
Allyl chloride	107-05-1
Arsenic	7440-38-2
Barium	7440-39-3
Benzene	71-43-2
Beryllium	7440-41-7
Bromodichloromethane [dichlorobromomethane]	75-27-4
Bromoform [tribromomethane]	75-25-2
Bromomethane [methyl bromide]	74-83-9
1,3-Butadiene	106-99-0
Cadmium	7440-43-9
Carbon disulfide	75-15-0
Carbon tetrachloride	56-23-5
Chlorobenzene	108-90-7
Chlorodibromomethane [dibromochloromethane]	124-48-1
Chloroform	67-66-3
Chloromethane [methyl chloride]	74-87-3
Chloroprene [2-chloro-1,3-butadiene]	126-99-8
Chromium VI	7440-47-3
Cobalt	7440-48-4
Cumene [isopropyl benzene]	98-82-8
Cyclohexanol	108-93-0
1,2-Dibromo-3-chloropropane	96-12-8
1,2-Dichlorobenzene [*o*-dichlorobenzene]	95-50-1
1,4-Dichlorobenzene [*p*-dichlorobenzene]	106-46-7
Dichlorodifluoromethane [CFC-12]	75-71-8
1,2-Dichloroethane [ethylene dichloride]	107-06-2
1,1-Dichloroethylene [vinylidene chloride]	75-35-4
1,2-Dichloropropane [propylene dichloride]	78-87-5
cis-1,3-Dichloropropylene	10061-01-5
trans-1,3-Dichloropropylene	10061-02-6
1,4-Dioxane [1,4-diethyleneoxide]	123-91-1
Epichlorohydrin [1-chloro-2,3-epoxypropane]	106-89-8
1,2-Epoxybutane	106-88-7
2-Ethoxyethanol [ethylene glycol monoethyl ether]	110-80-5
2-Ethoxyethanol acetate [2-EEA]	111-15-9
Ethylbenzene	100-41-4
Ethylene dibromide [1,2-dibromoethane]	106-93-4

(continued)

12

Table 1-1. (continued)

Constituent	CAS No.
Ethylene oxide	75-21-8
Formaldehyde	50-00-0
Furfural	98-01-1
Hexachloroethane	67-72-1
n-Hexane	110-54-3
Lead	7439-92-1
Manganese	7439-96-5
Mercury	7439-97-6
Methanol	67-56-1
2-Methoxyethanol	109-86-4
2-Methoxyethanol acetate [2-MEA]	110-49-6
Methyl tert-butyl ether	1634-04-4
Methylene chloride [dichloromethane]	75-09-2
Methyl ethyl ketone [2-butanone][MEK]	78-93-3
Methyl isobutyl ketone [hexone] [4-methyl-2-pentanone]	108-10-1
Methyl methacrylate	80-62-6
Naphthalene	91-20-3
Nickel	7440-02-0
2-Nitropropane	79-46-9
N-Nitrosodi-*n*-butylamine	924-16-3
N-Nitrosodiethylamine	55-18-5
N-Nitrosopyrrolidine	930-55-2
Propylene oxide	75-56-9
Pyridine	110-86-1
Styrene	100-42-5
1,1,1,2-Tetrachloroethane	630-20-6
1,1,2,2-Tetrachloroethane	79-34-5
Tetrachloroethylene [perchloroethylene]	127-18-4
Toluene	108-88-3
1,1,1-Trichloroethane [methyl chloroform]	71-55-6
1,1,2-Trichloroethane [vinyl trichloride]	79-00-5
Trichloroethylene	79-01-6
Trichlorofluoromethane [trichloromonofluoromethane]	75-69-4
1,1,2-Trichloro-1,2,2-trifluoroethane [freon 113]	76-13-1
Triethylamine	121-44-8
Vanadium	7440-62-2
Vinyl acetate	108-05-4
Vinyl chloride	75-01-4
Xylenes, mixed isomers [xylenes, total]	1330-20-7

Table 1-2. Constituents Modeled for Tanks Only

Constituent	CAS No.
Acrylamide	79-06-1
Acrylic acid	79-10-7
Aniline	62-53-3
Benzidine	92-87-5
Benzo(*a*)pyrene	50-32-8
2-Chlorophenol [*o*-chlorophenol]	95-57-8
Cresols, total	1319-77-3
7,12-Dimethylbenz[*a*]anthracene	57-97-6
N,N-Dimethyl formamide	68-12-2
3,4-Dimethylphenol	95-65-8
2,4-Dinitrotoluene	121-14-2
1,2-Diphenylhydrazine	122-66-7
Ethylene glycol	107-21-1
Hexachlorobenzene	118-74-1
Hexachloro-1,3-butadiene [hexachlorobutadiene]	87-68-3
Hexachlorocyclopentadiene	77-47-4
Isophorone	78-59-1
3-Methylcholanthrene	56-49-5
Nitrobenzene	98-95-3
Phenol	108-95-2
Phthalic anhydride	85-44-9
2,3,7,8-TCDD [2,3,7,8-tetrachlorodibenzo-*p*-dioxin]	1746-01-6
o-Toluidine	95-53-4
1,2,4-Trichlorobenzene	120-82-1

Emissions, transport, and exposure were modeled somewhat differently for the three risk endpoints (chronic, subchronic, and acute). For emissions and transport, different averaging times were used for each endpoint (1 year for chronic, 1 month for subchronic, and 1 day for acute) to generate emission rates and dispersion factors. For exposure, subchronic and acute exposures were modeled deterministically, using the point of maximum exposure at a specific distance. Chronic exposures were modeled probabilistically using a Monte Carlo approach to capture variation in receptor location and exposure factors. The WMUs assessed are aerated treatment tanks, nonaerated treatment tanks, storage tanks, landfills, waste piles, and land application units. The risk assessment was structured to capture national variations in environmental settings. In addition, Monte Carlo analysis was used in the modeling to include the variations in receptor characteristics such as exposure parameters and location around the facility.

14

1.3 Organization of Report

The remainder of this report is organized as follows. Section 2 summarizes changes made from the 1998 Air Characteristic Study. Section 3 provides a general overview of the risk analysis. Section 4 presents the revised risk analysis results. Section 5 presents the integration of the revised risk assessment results with the May 1998 regulatory gaps and occurrence analyses. References are provided in Section 6, and supporting analyses are included in Appendix A.

1.4 Companion Documents

Volume II of this report, the *Technical Background Document*, provides a detailed description of the methodologies, data, and supporting analyses used for the risk assessment.

Volume III of this report, *Results* (provided on CD-ROM), presents the detailed results of the risk analysis.

15

2.0 Revisions to the Risk Assessment Framework

The analytical approach used for this analysis differs in important ways from the approach used for the May 1998 Air Characteristic Study. Changes have been made to the risk assessment to improve the robustness of the analysis, reduce uncertainty, and make corrections to the 1998 study. The changes reflect comments made by peer-reviewers and public commenters and other improvements made by the Agency. Several aspects of the Air Characteristic Study risk assessment were modified, including source characterization, emissions modeling, air dispersion modeling, health benchmarks, and exposure and risk modeling. These changes are discussed in the following sections.

2.1 Source Characterization

Several changes were made to improve the source characterization. The source characterization is the information about waste management unit (WMU) dimensions and operations that defines how Industrial D waste is managed. It is important to accurately establish these characteristics since they influence the rate of emissins and amount of dispersion of a constituent.

Changes in source characterizations have affected all the WMU categories. These changes are discussed for each WMU in the following sections. Except for tanks, source characterizations were and still are based on the Subtitle D Survey (Schroeder et al., 1987); however, the actual number of units included in the analysis from that survey has increased slightly. Those increases are also discussed in the following sections.

2.1.1 Landfills

In both the May 1998 study and the current study, landfills were modeled assuming that the landfill is divided into an equal number of cells. The cell size is determined by dividing the total landfill area by the landfill life, 20 years, creating 20 cells for each landfill. One cell operates for 1 year for the life of the landfill. Each cell is assumed to be covered at the end of the year, preventing further emissions from that cell. Thus, emissions are only occurring from one cell at any given time, or from an area equal to one-twentieth of the total area. Emissions from the open cell are modeled as an emission rate per unit area (g/m^2-s), and total emissions in g/s are then calculated by multiplying this per-unit-area emission rate by the area from which emissions occur. In the May 1998 study, the total area, instead of the cell area, was used to calculate total emissions. This error has been corrected in the current study. This correction reduces total emissions by a factor of 20 and air concentration (and therefore risk) by about a factor of 10 (because dispersion is not linear on source area). As a result of this change, the protective waste concentrations for landfills are higher by about a factor of 10.

116

In the May 1998 study, 790 landfills were modeled from the Subtitle D survey data. This reflected a total of 827 landfills reported, 37 of which were culled for various reasons. Eleven of the 37 were culled based on results of previous groundwater modeling work by EPA not related to the Air Characteristic Study. Those 11 sites are, however, relevant to the Air Characteristic Study. Because retaining as many sites as possible is desirable, those 11 sites were included this time, resulting in a database of 801 landfills for the current study. The culls made for the current study are detailed in Volume II, Section 3.1.

2.1.2 Land Application Units

The Subtitle D survey does not provide data on application frequency for land application units (LAUs). In the May 1998 study, application frequency was assumed to be four times per year. Sensitivity analysis shows that the application frequency has a significant impact on emissions even when total annual waste quantity is held constant: the more frequent the applications, the greater the emissions. This is due in part to the fact that tilling is presumed to occur whenever waste is applied, and tilling increases emissions by disturbing the waste. Therefore, we reviewed several data sources in an effort to better characterize application frequency for LAUs. These sources included *Land Treatment Practices in the Petroleum Industry* (Environmental Research & Technology, 1983); *Review and Evaluation of Current Design and Management Practices for Land Treatment Units Receiving Petroleum Wastes* (Martin et al., 1986); and *Handbook of Land Treatment Systems for Industrial and Municipal Wastes* (Reed and Crites, 1984). Data in these sources were used to establish a relationship between the number of applications per year and the annual waste quantity managed. This relationship was applied to the LAUs in the Subtitle D survey to establish a distribution of application frequencies relevent to Industrial D LAUs. An application frequency of 24 times per year was selected for use in this study, reflecting a central tendency value from the distribution (see Volume II, Section 4.5.1 for more details).

The increase in application frequency from 4 applications per year to 24 applications per year should increase emissions (and therefore risk) and decrease the protective waste concentration.

In the May 1998 study, 308 land application units were modeled from the Subtitle D survey data. This reflected a total of 354 land application units reported, 46 of which were culled for various reasons. Thirty-seven of the sites culled were culled based on previous groundwater modeling work by EPA not related to the Air Characteristic Study. Those 31 sites are relevant to the Air Characteristic Study. Because retaining as many sites as possible is desirable, those sites were included this time. Many of these sites had been culled because the reported waste quantity and area implied an unrealistically large application rate (greater than 10,000 tons/acre/yr). Those sites were retained in this study, and new waste quantities were imputed that did not violate this criterion (see Volume II, Section 3.1.3 for more details). This resulted in a database of 345 land application units for the current study. The culls made for the current study are detailed in Volume II, Section 3.1.

2.1.3 Wastepiles

The Subtitle D survey does not provide data on wastepile height. In the May 1998 study, wastepiles were modeled based on two assumed heights (2 and 5 meters). However, wastepile height is related to wastepile area, waste quantity, and retention time. Therefore, it is more realistic to evaluate the characteristics of each wastepile and assign a height individually. Wastepile area and waste quantity are reported in the Subtitle D survey, but retention time is not. Therefore, to tailor the wastepile heights to known data, a relationship between wastepile area, waste quantity, and height was developed (see Volume II, Section 3.1.1 for more details). In this study, each wastepile modeled was assigned a height based on that relationship. Height is used only in the dispersion modeling and affects air concentration; the greater the height, the greater the dispersion, and so the lower the air concentration at a particular location. The dispersion model is not sensitive to small changes in height; therefore, to simplify dispersion modeling without sacrificing accuracy, a set of six discrete heights, covering the range of heights calculated for all the wastepiles, was used. These heights were 1, 2, 4, 6, 8, and 10 meters.

Approximately 77 percent of the wastepiles modeled were assigned a height of 1 m, and 95 percent of the wastepiles were assigned a height of 4 m or less. Therefore, most of the wastepiles are modeled at a lower height in the current study than in the May 1998 study. As a result, air concentration (and therefore risk) will tend to be greater and the protective waste concentration lower. This difference is significant: about a factor of 2 to 10 relative to the 2-m wastepiles in the May 1998 study and a factor of 2 to 25 relative to the 5-m wastepiles in the May 1998 study.

In the 1998 study, 742 wastepiles were modeled from the Subtitle D survey data. This reflected a total of 853 wastepiles reported, of which 111 were culled for various reasons (most because they are Bevill facilities, which are exempt from Subtitle C regulation and would therefore never be subject to an Air Characteristic under Subtitle C). Three of the sites culled were culled based on previous groundwater modeling work by EPA not related to the Air Characteristic Study. Those three sites are relevant to the Air Characteristic Study. Because retaining as many sites as possible is desirable, those three sites were included this time, resulting in a database of 745 wastepiles for the current study. The culls made for the current study are detailed in Volume II, Section 3.1.

2.1.4 Tanks

The tank source category has been revised extensively, with respect to both how tanks are characterized and the categories of tanks modeled.

Because the Subtitle D survey did not contain data on tanks, they were characterized in the May 1998 study using two model tanks placed at 29 locations. A full distribution of tanks, using a database of many actual tank facilities (as was done for the other WMUs), would provide a more representative result. However, data on Industrial D tanks do not exist; therefore, in the current study, tanks were characterized using tank data from the 1986 National Survey of Hazardous Waste Treatment, Storage, Disposal, and Recycling Facilities (TSDR) Database (U.S. EPA, 1987) (see Volume II, Section 3.4 for more details). These data provide a distribution of tanks to represent the range of tank configurations used in the United States. This is an important

18

step in improving the analysis because some tank characteristics are critical parameters in the emissions modeling. However, the TSDR tank database did not include data on all parameters needed for the emission and dispersion modeling; therefore, data from site visits to tanks done by the Agency in 1985 and 1986 in support of the development of RCRA Air Emission Standards were used to develop some of the tank-specific parameter values to characterize tank engineering and operating parameters (see Volume II, Section 3.4.2 for more details). This introduces some uncertainty into the tank characterization; however, we believe this uncertainty to be less than the uncertainy arising from the use of only two model tanks to characterize the universe of Industrial D tanks.

In the May 1998 study, four categories of tanks were modeled: aerated tanks with and without biodegradation and storage tanks with and without biodegradation. These categories do not capture nonaerated treatment tanks. Based on the TSDR tank data, nonaerated treatment tanks appear to differ from storage tanks (which are also nonaerated) in important ways, particularly with respect to the distribution of area. In addition, storage tanks are not designed for biodegradation, so the category of storage tank with biodegradation is not representative of real tanks. The tank categories modeled in the current study include aerated treatment tanks, nonaerated treatment tanks, and storage tanks. Some of the aerated treatment tanks were modeled with biodegradation, while others were modeled without biodegradation, depending on the treatment process reported. Nonaerated treatment tanks, like storage tanks, are typically not optimized for biodegradation; therefore, both nonaerated treatment tanks and storage tanks were modeled with no biodegradation.

Figure 2-1 shows the distribution of area for the three tank categories modeled in this study and shows where the two model tanks used in the 1998 study fall relative to those distributions. Table 2-1 shows exactly where the two model tanks fall in the new tank distributions. Because the two model tanks both fall relatively high in the new distributions, the use of the new distributions will tend to decrease tank size (and therefore risk) and increase the 90^{th} percentile protective waste concentration. The elimination of biodegradation from many of the tanks will increase emissions (and therefore risk) and tend to decrease the protective waste concentration relative to tank types with biodegradation from the 1998 study.

2.2 Emissions Modeling

Changes in emissions modeling have affected all of the land-based WMUs. Tank emissions modeling was not changed; the new distribution of tanks is modeled in the same manner as in the May 1998 study with regard to emissions estimates. Changes in emissions modeling for land-based units were considered for both volatile and particulate emissions.

For particulate emissions, most of the parameter values used to compute particulate emission rates are site-specific. Some vary with the waste (e.g., silt content of the waste), others with the waste management unit (e.g., roughness height of unit, vegetative cover on unit), and still others with the location of the unit (e.g., meteorological parameters like precipitation data and windspeed). The 1998 study did account for the variability of the meteorological parameters by varying these based on assigned location of the unit; however, variability in the other parameters was not captured. For the current study, consideration was given to developing distributions of the other parameter values in the particulate emissions model for use in the

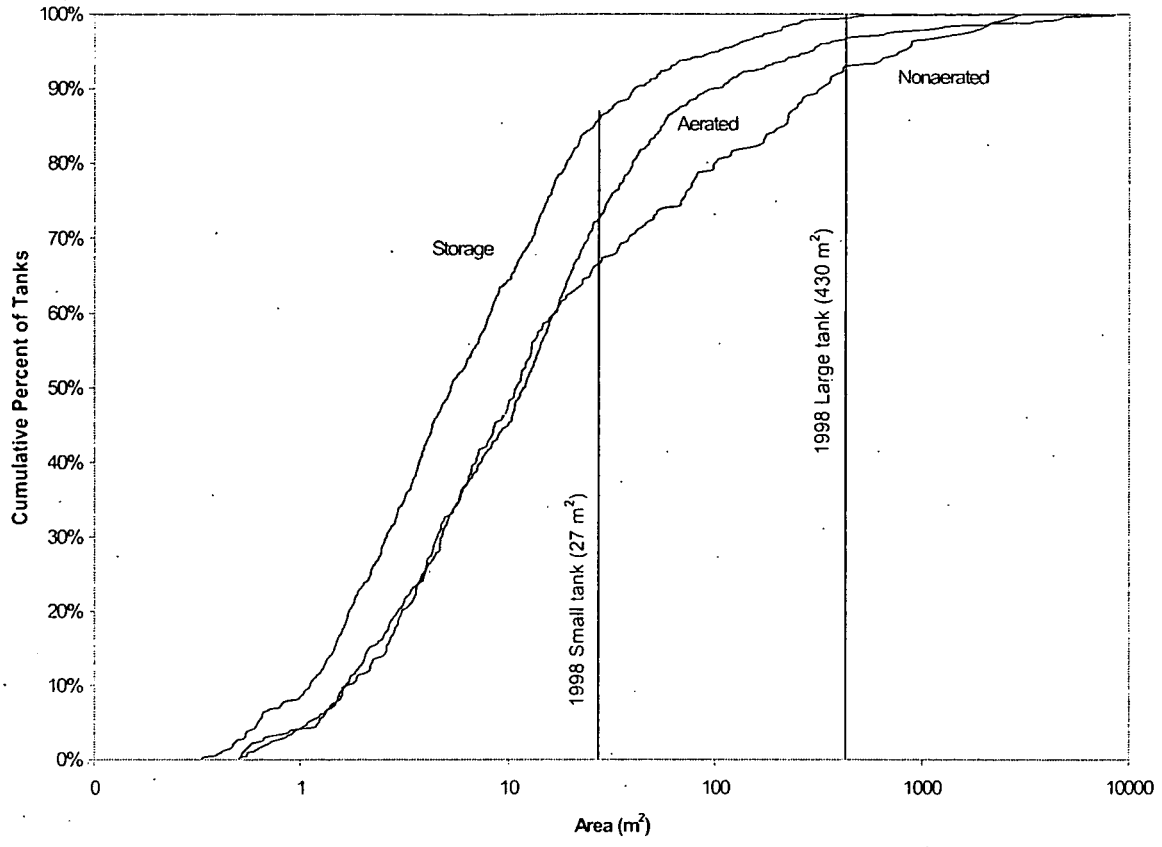


Figure 2-1. Cumulative tank distributions used in the current study.

Table 2-1. Percentiles of New Tank Distributions Associated with Two Model Tanks

	Percent of new tanks that are smaller than the model tank:	
Type of Tank	**Small model tank (27 m²)**	**Large model tank (430 m²)**
Aerated treatment	73%	97%
Nonaerated treatment	66%	93%
Storage	86%	100%

20

Monte Carlo model, in order to capture variability in the protective waste concentration due to variation in these parameters and to provide a more complete description of the variability of the protective waste concentrations. However, no data were identified that would support development of distributions, so no change was made to the particulate emissions modeling. The exclusion of such distributions only affects the distribution of results for metals; particulate emissions for volatile constituents are trivial compared to volatile emissions, so such a refinement would not have a significant effect on the results for volatile constituents.

Modeling of volatile emissions from all land-based units was modified with respect to the treatment of adsorbtion. Adsorption/absorption is the tendency of a chemical or liquid media to attach or bind to the surface or fill the pores of particles in the soil or waste and therefore not volatilize into the air. This tendency of a chemical to adsorb to or absorb in particles is important to consider in estimating the concentration of the chemical on particles emitted to the air due to wind erosion. The CHEMDAT8 model estimates emissions from land-based WMUs using a simple emissions model that accounts for contaminant partitioning between a liquid waste matrix and the air, diffusion of vapors through a porous media, and contaminant loss through biodegradation. This model accounts for adsorbtion when the waste concentration entered is a liquid-phase concentration; however, it does not account for adsorbtion when a total waste concentration (i.e., liquid and solid phase) is entered. The assumption of an entered waste concentration in liquid phase was based on the petroleum wastes for which CHEMDAT8 was originally developed and may not apply to the chemicals considered in this analysis. Therefore, a method for including adsorptive partitioning for total waste concentrations was developed and used to modify CHEMDAT8 for the current study. The changes to the CHEMDAT8 code are shown in detail in Section 4.3 of Volume II. This change should tend to decrease emissions and risk and increase protective waste concentration; the extent of the decrease in emissions will be constituent-specific, depending on the constituent's tendency to adsorb to particles.

The LAU emissions model was changed substantially. Instead of quarterly meteorological data, which were used in the May 1998 study, monthly meteorological data were used. Monthly meteorological data are more consistent with the application rate used in this study (24 applications per year). In addition, changes were made to the approach for estimating long term emission rates for LAUs in the current study. The May 1998 version of the model used steady state assumptions to estimate long-term emission rates. This presumed that all chemicals reached steady state emissions immediately (i.e., concentration remaining in the unit remains constant over time because waste additions and losses balance each other). While most chemicals will reach steady state within 1 or 2 years, some chemicals take longer than that or may never reach steady state. In order to better address the time to reach steady state in the current study, emissions were estimated using a pseudo-steady state approach, in which a series of steady state solutions was calculated for many short time periods, and the resulting emission rates were averaged to estimate long-term emissions. Specifically, emissions were estimated on a monthly basis for 40 years, and monthly emission rates for year 40 were averaged to estimate long-term annual emission rates. The actual length of time to reach steady state is constituent-specific. Forty years was chosen as a sufficiently long time for all chemicals that would ever reach steady state to do so. For those constituents that reach steady state sooner, there is no difference between using the first year after steady state is reached (typically year 2 or 3) and year 40. However, for constituents that do take many years to reach steady state, this approach provides a more realistic estimate of emissions.

21

An error in soil biodegradation rates that affects chronic, subchronic, and acute volatile emission rates for LAUs and chronic volatile emissions for wastepiles was identified and corrected for the current study (the differences between chronic, subchronic, and acute are discussed in Section 3.1). In the May 1998 study, the soil biodegradation rates were erroneously labeled as half lives and therefore used incorrectly. While half life is related to the first-order biodegradation rate, the two are not interchangeable. When this error was discovered, all soil biodegradation rates were verified against the original data source (Howard et al., 1991). The current study correctly uses the verified biodegradation rates from Howard et al. (1991). The impact of this error is chemical-specific. Table 2-2 summarizes the direction of the error for the chemicals modeled in land-based units. For about half of these, the biodegradation rate was too low, resulting in an overestimate of emissions. Emissions modeled with the correct biodegradation rate will be lower, resulting in less risk and a higher protective waste concentration. Most of the remaining chemicals were relatively unaffected by this correction. For only one chemical was the incorrect biodegradation rate too high, resulting in an underestimate of emissions. The corrected emissions for this chemical will be higher, resulting in more risk and a lower protective waste concentration.

2.3 Air Dispersion Modeling

Several changes in dispersion modeling were implemented in the current study and make the dispersion modeling more accurate than in the May 1998 study.

In the May 1998 study, wet and dry depletion of the atmospheric concentrations (plume depletion) of vapors and particulates were not considered due to the great increase in run time of ISCST3 for area sources when depletion is modeled (run times with depletion for area sources are typically 15 to 30 times longer than run times without depletion) and the short timeframe for completing the study. However, plume depletion can have a significant effect on air concentration, especially for particulates, and many of the peer-review comments identified this as a serious shortcoming of the May 1998 study. Therefore, for this analysis, with more time available, the issue of depletion was revisited. In addition, since May 1998, it had come to light for other EPA work (the Hazardous Waste Identification Rule, or HWIR) that the precipitation data in the hourly meteorological data used in the dispersion model were incomplete (i.e., some hours had missing precipitation data, resulting in total precipitation less than actual precipitation), which could affect the amount of wet depletion occurring. Work had already been done for the HWIR project to interpolate missing precipitation data for many of the meteorological locations modeled in the Air Characteristic Study.

Several sensitivity analyses were conducted to assess the importance of wet and dry depletion for particulates and wet depletion for vapors (ISCST3 cannot model dry depletion of vapors; therefore, this could not be considered. However, dry depletion of vapors is expected to be negligible). These sensitivity analyses showed that for both vapors and particulates, wet depletion did not have a significant impact on air concentrations (differences were less than 2 percent), even using the more complete interpolated precipitation data developed for HWIR. Dry depletion of particulates, on the other hand, did have a significant effect on air concentration (differences ranged up to about 40 percent). Therefore, dry depletion of particles was included in the current study for all land-based units. This change will reduce the air concentration of

22

Table 2-2. Chemical-specific Effects of Biodegradation Rate Correction

CAS	Chemical	New soil biodegradation rate (sec^{-1})	Old soil biodegradation rate (sec^{-1})
Higher emissions, lower C_w			
78875	Dichloropropane, 1,2-	6.2E-09	1.1E-07
Lower emissions, higher C_w			
75070	Acetaldehyde	1.1E-06	1E-20
67641	Acetone	1.1E-06	6.1E-10
75058	Acetonitrile	2.9E-07	2.4E-09
107028	Acrolein	2.9E-07	2.4E-09
107131	Acrylonitrile	3.5E-07	2.0E-09
107051	Allyl chloride	5.7E-07	1.2E-09
71432	Benzene	5.0E-07	1.4E-09
75274	Bromodichloromethane	4.5E-08	1E-20
106990	Butadiene, 1,3-	2.9E-07	1E-20
67663	Chloroform	4.5E-08	2.4E-09
98828	Cumene	1.0E-06	7.0E-10
108930	Cyclohexanol	4.5E-08	1E-20
106467	Dichlorobenzene, p-	4.5E-08	1E-20
10061015	Dichloropropylene, cis-1,3-	7.1E-07	9.8E-10
10061026	Dichloropropylene, trans-1,3-	7.1E-07	9.8E-10
106898	Epichlorohydrin	2.9E-07	2.4E-09
106887	Epoxybutane, 1,2-	6.2E-07	1E-20
111159	Ethoxyethanol acetate, 2-	2.9E-07	1E-20
110805	Ethoxyethanol, 2-	2.9E-07	2.4E-09
100414	Ethylbenzene	8.0E-07	8.7E-10
75218	Ethylene oxide	6.8E-07	1E-20
50000	Formaldehyde	1.1E-06	6.1E-10
67561	Methanol	1.1E-06	6.1E-10
110496	Methoxyethanol acetate, 2-	2.9E-07	1E-20
109864	Methoxyethanol, 2-	2.9E-07	1E-20
74839	Methyl bromide	2.9E-07	2.4E-09
74873	Methyl chloride	2.9E-07	2.4E-09
78933	Methyl ethyl ketone	1.1E-06	6.1E-10
108101	Methyl isobutyl ketone	1.1E-06	6.1E-10
80626	Methyl methacrylate	2.9E-07	2.4E-09
1634044	Methyl tert-butyl ether	4.5E-08	1E-20
75092	Methylene chloride	2.9E-07	2.4E-09
91203	Naphthalene	1.7E-07	4.2E-09
110543	n-Hexane	5.0E-07	1E-20
924163	N-Nitrosodi-n-butylamine	4.5E-08	1E-20
75569	Propylene oxide	6.5E-07	1E-20
110861	Pyridine	1.1E-06	6.1E-10
100425	Styrene	2.9E-07	2.4E-09
630206	Tetrachloroethane, 1,1,1,2-	1.8E-07	5.8E-09
79345	Tetrachloroethane, 1,1,2,2-	1.8E-07	3.8E-09
108883	Toluene	3.6E-07	1.9E-09

(continued)

Table 2-2. (continued)

CAS	Chemical	New soil biodegradation rate (sec^{-1})	Old soil biodegradation rate (sec^{-1})
76131	Trichloro-1,2,2-trifluoroethane, 1,1,2-	2.2E-08	1E-20
108054	Vinyl acetate	1.1E-06	1E-20
1330207	Xylenes	2.9E-07	2.4E-09
Relatively unaffected			
7440382	Arsenic	0	1E-20
7440393	Barium	0	1E-20
7440417	Beryllium	0	1E-20
7440439	Cadmium	0	1E-20
75150	Carbon disulfide	0	1E-20
56235	Carbon tetrachloride	2.2E-08	3.1E-08
108907	Chlorobenzene	5.3E-08	1.3E-08
124481	Chlorodibromomethane	4.5E-08	1.6E-08
126998	Chloroprene	4.5E-08	1.6E-08
7440473	Chromium (total)	0	1E-20
7440484	Cobalt	0	1E-20
96128	Dibromo-3-chloropropane, 1,2-	4.5E-08	1.6E-08
95501	Dichlorobenzene, o-	4.5E-08	1.6E-08
75718	Dichlorodifluoromethane	4.5E-08	1.6E-08
107062	Dichloroethane, 1,2-	4.5E-08	1.6E-08
75354	Dichloroethylene, 1,1-	4.5E-08	1.6E-08
123911	Dioxane, 1,4-	4.5E-08	1.6E-08
106934	Ethylene dibromide	4.5E-08	1.6E-08
98011	Furfural	0	1E-20
67721	Hexachloroethane	4.5E-08	1.6E-08
7439921	Lead	0	1E-20
7439965	Manganese	0	1E-20
7439976	Mercury	0	1E-20
7440020	Nickel	0	1E-20
79469	Nitropropane, 2-	4.5E-08	1.6E-08
55185	N-Nitrosodiethylamine	4.5E-08	1.6E-08
930552	N-Nitrosopyrrolidine	4.5E-08	1.6E-08
127184	Tetrachloroethylene	2.2E-08	3.1E-08
75252	Tribromomethane	4.5E-08	1.6E-08
71556	Trichloroethane, 1,1,1-	2.9E-08	2.4E-08
79005	Trichloroethane, 1,1,2-	2.2E-08	3.2E-08
79016	Trichloroethylene	2.2E-08	3.1E-08
75694	Trichlorofluoromethane	2.2E-08	3.1E-08
121448	Triethylamine	0	1E-20
7440622	Vanadium	0	1E-20
75014	Vinyl chloride	4.5E-08	1.6E-08

24

particulates, reducing risk and increasing the protective waste concentration for land-based units. The change is only significant for metals, however, because particulate emissions of volatile constituents are negligible compared to volatile emissions.

As discussed earlier, several changes to the source characterization were made that required new dispersion modeling. These changes included the addition of more specific heights for wastepiles and the recharacterization of tanks.

Dispersion modeling for wastepiles was modified to better capture the effect of wastepile height on ground-level concentrations. Section 2.1.3 discusses these changes. The difference in the results is significant—about a factor of 2 to 10 relative to the 2-m wastepiles in the May 1998 study and a factor of 2 to 25 relative to the 5-m wastepiles in the May 1998 study.

Dispersion modeling for tanks was also modified to capture the range of area/height combinations reflected in the new tank characterizations. A total of 33 area/height combinations were modeled for tanks, compared to only 2 area/height combinations (corresponding to the two model tanks) in the May 1998 study. The overall effect of the new area-height combinations compared to the ones used last year is not clearly in one direction, as the effect of the change in areas has effects in the opposite direction of the effect of the change in heights. As shown in Figure 2-1, the two model tanks used in the May 1998 study fall fairly high on the distribution of tanks used in the current study. Therefore, many of the tanks modeled in the current study are smaller in area, which will tend to result in lower air concentrations, lower risk, and higher protective waste concentrations relative to the May 1998 study. However, the two heights modeled in the May 1998 study were also high relative to the distribution of heights modeled in the current study, which has the opposite effect: the generally lower heights will tend to increase air concentration and risk, and lower protective waste concentration.

Finally, an error in the interpolation of dispersion coefficients for wastepiles in the May 1998 study was discovered and corrected for the current study. In the May 1998 study, the areas used for the interpolation were those for tanks, not wastepiles, which resulted in interpolated UACs (and therefore air concentration and risk) that are too low by a factor of 2 to 3.

2.4 Human Health Benchmarks

Twenty-eight of the inhalation benchmarks used in the May 1998 study have been changed. The changes are summarized in Table 2-3. More detailed information is available in Volume II, Section 6.0.

In some cases, new IRIS or other published information became available during the past year that suggested a change in the inhalation benchmark for this study.

The progression of values from chronic to subchronic to acute benchmarks was also reviewed, especially when the chronic value exceeded the subchronic or acute value. The anticipated progression would reflect that high concentrations of a chemical can be tolerated without ill effect for shorter periods of exposure than for longer periods of exposure. Therefore,

Table 2-3. Summary of Issues and Changes for Inhalation Benchmarks Used in the Air Characteristic Study

CAS	Name	Issue	Resolution
75-05-8	Acetonitrile	New IRIS RfC=0.06 mg/m^3, appropriate to use as subchronic	Revise chronic and subchronic RfCs
7440-38-2	Arsenic	CalEPA acute REL updated	Revise acute RfC
75-15-0	Carbon disulfide	Chronic RfC target organ should be neurological; CalEPA acute REL updated	Revise chronic RfC target organ and acute RfC
7440-47-3	Chromium VI	New IRIS RfC= 1E-4 mg/m^3 (particulates); revised intermediate MRL= 5E-4 mg/m^3 (particulates)	Revise chronic and subchronic RfCs
1319-77-3	Cresols (total)	Received public comment on chronic RfC; subchronic lower than chronic (due to calculation error)	No revision on chronic RfC; revise subchronic RfC (1.2E-3 mg/m^3)
108-93-0	Cyclohexanol	New FR RfC=2E-5 mg/m^3	Revise chronic RfC; recalculate subchronic (2E-4 mg/m^3)
106-46-7	Dichlorobenzene, 1,4-	Chronic RfC target organ should be liver	Revise chronic RfC target organ
107-06-2	Dichloroethane, 1,2-	Acute RfC lower than subchronic	Revise subchronic RfC (acute MRL = chronic MRL, therefore should also = subchronic = 0.81 mg/m^3)
123-91-1	Dioxane, 1,4-	Acute RfC lower than subchronic; CalEPA acute REL updated	Update acute RfC (still incorrect progression - see text)
106-89-8	Epichlorohydrin	CalEPA acute REL updated	Revise acute RfC
111-15-9	Ethoxyethanol acetate, 2-	CalEPA acute REL updated	Revise acute RfC
110-80-5	Ethoxyethanol, 2-	Acute RfC lower than subchronic; CalEPA acute REL updated; chronic RfC target organ should be male reproductive and hematological	Update acute RfC (still incorrect progression - see text); revise chronic RfC target organ
78-59-1	Isophorone	New FR RfC=1.2E-2 mg/m^3	Revise chronic RfC; recalculate subchronic (1.2E-1 mg/m^3)
7439-97-6	Mercury	CalEPA acute REL updated	Revise acute RfC
67-56-1	Methanol	CalEPA acute REL updated	Revise acute RfC
109-86-4	Methoxyethanol, 2-	Acute RfC lower than subchronic; CalEPA acute REL updated	Update acute RfC (still incorrect progression - see text)
78-93-3	Methyl ethyl ketone	CalEPA acute REL updated	Revise acute RfC
75-09-2	Methylene chloride	Revised acute MRL= 3 ppm (10 mg/m^3)	Revise acute RfC
91-20-3	Naphthalene	New IRIS RfC= 3E-3 mg/m^3	Revise chronic RfC; recalculate subchronic RfC (3E-2 mg/m^3)
7440-02-0	Nickel	CalEPA acute REL updated	Revise acute RfC
108-95-2	Phenol	Public comment on chronic RfC; new FR RfC=6E-3 mg/m^3; CalEPA acute REL updated	Revise chronic RfC as per FR; recalculate subchronic RfC (6E-2 mg/m^3); revise acute RfC
75-56-9	Propylene oxide	CalEPA acute REL updated	Revise acute RfC
100-42-5	Styrene	CalEPA acute REL updated	Revise acute RfC
1746-01-6	TCDD, 2,3,7,8-	URF available (3.3E+1 per µg/m^3) in HEAST	Add URF
127-18-4	Tetrachloroethylene	Acute RfC lower than subchronic; cancer benchmarks available - Superfund URF (5.8E-7 per µg/m^3) and CSF (2E-3 per mg/kg/d)	No revision of acute RfC; revise URF & CSF
108-88-3	Toluene	Revised acute MRL = 4 ppm (15 mg/m^3)	Revise acute RfC

(continued)

26

Table 2-3. (continued)

CAS	Name	Issue	Resolution
7440-62-2	Vanadium	Public comment on chronic RfC; acute RfC lower than subchronic	Revise subchronic RfC (subchronic = chronic = 7E-5 mg/m^3); recalculate acute (7E-4 mg/m^3)
1330-20-7	Xylenes (total)	Calculation error for chronic RfC	Revise chronic RfC (4E-1 mg/m^3)

CSF = cancer slope factor
FR = *Federal Register*
MRL = minimal risk level
REL = reference exposure level
RfC = reference concentration
URF = unit risk factor

chronic noncarcinogenic benchmarks should be lower than subchronic benchmarks, and subchronic benchmarks should be lower than acute benchmarks (note that for chronic benchmarks, this comparison can only be meaningfully made for noncarcinogens, since the chronic carcinogenic slope factor cannot be directly compared to a subchronic or acute benchmark). This review resulted in some modifications; however, data were not available to correct all instances in which the progression from chronic to subchronic to acute was not as expected. Subchronic and acute benchmarks are typically obtained from different sources and based on different underlying studies than chronic benchmarks are. Inconsistencies in how the benchmarks were developed or the underlying studies used often accounts for the discrepancy in expected progression. In many cases, no set of benchmarks could be found that displayed the expected progression.

Finally, public commenters on the May 1998 study specifically identified benchmarks for four constituents that should be reviewed: cobalt, cresols, phenol, and vanadium. These reviews resulted in changes as well.

In addition to changes in the basic health benchmarks described above, a change was made in how the cancer slope factors were used for children. Slope factors are developed for adults, using an assumed body weight of 70 kg. In the May 1998 study, slope factors were adjusted for children based on actual body weight. However, based on peer-review comments and further discussion with Agency experts in cancer dose-response, this adjustment has been eliminated from the current study. Cancer slope factors are used as presented for children, without adjustment. It should be noted, however, that the differences between an adult's and a child's physiology are accounted for in this study by adjusting the appropriate exposure factors. This is discussed in the following section.

2.5 Exposure and Risk Modeling

Four changes were made to the exposure and risk model used in this analyis:

Update of the exposure factor distributions
New child exposure approach

27

Change in worker scenarios
Change in checks for nonlinearity.

Exposure factors used in this study include body weight, inhalation rate, and exposure duration. Distributions of these exposure factors were updated from those used in the May 1998 study. Previously, data for males were used. However, this does not account for differences between males and females. Males typically have higher body weights and inhalation rates than females, as well as higher inhalation rates per unit of body weight. For this update, data on both males and females were used to better capture the potential effects on the whole population, not just males. Because females have a lower inhalation rate per unit of body weight than males, the effect should be to lower the overall distribution of risk and increase the protective waste concentration.

Three child age groups, or cohorts, were used to model child exposures: 0 to 3, 4 to 10, and 11 to 18 years of age. These cohorts are unchanged from the May 1998 study and reflect the age cohorts for which inhalation rate data are available. In the May 1998 study, the results were presented as a single "child" receptor. For each iteration of the Monte Carlo analysis, a starting age for exposure was chosen at random from among 0, 4, and 11 years (the three cohort starting ages), with the probability of each of the three starting ages being chosen proportional to the total number of years in the cohort. A single exposure duration was associated with each of the three starting ages, based on the median for that age cohort, and this was not varied for a particular starting age. None of the three exposure durations, when coupled with their associated starting age, resulted in exposure past age 18. This approach does not fully capture the impact of different starting ages (since these were restricted to three) or the full variability of exposure duration.

For this study, the child exposure approach was modified to better capture variations in age at start of exposure and exposure duration. Results were calculated and saved separately for receptors falling into each of the three age cohorts at the start of exposure (note that exposure may last longer than just the range of ages in a cohort); these are presented as "child 0-3 years," "child 4-10 years," and "child 11-18 years." For each iteration for a cohort, exposure begins at a starting age selected at random within the cohort (with each year of age within the cohort having equal probability of being selected). An exposure duration is also selected at random for each iteration from a distribution for the cohort (so there are three exposure duration distributions, one per cohort). Exposure is then started at the selected starting age and continues through succeeding age cohorts as necessary until the exposure duration selected for that starting age is reached. Depending on the starting age and exposure duration selected, exposure may continue into adulthood.

Both on-site workers and off-site workers were included in the May 1998 study. For this study, the on-site workers are no longer included in the receptors modeled. Accordingly, because concentration at 0 m from the WMU was only used for the on-site worker scenario, it has been dropped. Off-site workers have been retained and are evaluated for all receptor locations, as before.

A units conversion error in the calculation of a hazard quotient for lead was identified in the May 1998 model and corrected for this study. Specifically, a units conversion factor to

26

convert air concentration from $\mu g/m^3$ to mg/m^3 was omitted. As a result, the hazard quotients for lead were too high by a factor of 1,000 and the protective waste concentrations too low by the same amount. Due to the modeling of different receptors for lead (children age 0 to 3 years and 3 to 7 years), the hazard quotient equation for lead was separate from the equation used for all other chemicals, so this error did not affect any other chemicals.

Finally, the approach to adjusting the results for nonlinearities in the emissions modeled has been changed. In the May 1998 study, two types of adjustments were made that have been dropped for the current study. These are discussed below.

In the May 1998 study, waste concentrations were backcalculated using both an aqueous-phase emission rate (modeled using Henry's law) and an organic-phase emission rate (modeled using Raoult's law). Typically, the aqueous-phase emission rates are much higher than the organic-phase emissions rates, resulting in lower waste concentrations based on aqueous-phase emission rates, but for a few chemicals that was not the case. In those cases, the backcalculated concentration was adjusted in the May 1998 study to be based on the organic-phase emission rate. However, the fact that greater emissions occur from the organic phase does not alter the fact that the aqueous phase is a far more likely scenario for the waste management units modeled in this study. The Agency decided that this adjustment was unnecessarily worst-case; therefore, this adjustment was dropped in the current study, and all results are based on the aqueous-phase emission rates. Results that would be lower if based on the organic-phase emission rates are footnoted.

In the May 1998 study, the backcalculated waste concentration based on the aqueous-phase emission rate was compared to either the soil saturation concentration (for land-based units) or the solubility at neutral pH and a temperature of 20-25 °C. These are the theoretical maximum concentrations at which aqueous phase wastes can exist; at higher concentrations, the waste is organic phase. If the backcalculated waste concentration based on aqueous-phase emission rates exceeded the soil saturation concentration or solubility, then it was adjusted to be based on the organic-phase emission rate instead. However, the soil saturation concentration and solubility are both dependent on site- and waste-specific conditions such as temperature and pH. Therefore, a backcalculated waste concentration near the soil saturation concentration or solubility calculated for this study may be possible in some situations and not in others. Rather than artificially restrict the results to standard conditions, in the current study all results are based on the aqueous-phase emission rates. If this backcalculated concentration exceeds the soil saturation concentration or solubility calculated for this study, the result is footnoted, and the footnote identifies whether pure organic-phase component (i.e., 1 million ppm modeled as organic phase) results in a risk greater or less than the cutoff risk of 10^{-5} or the cutoff HQ of 1.

29

3.0 Summary of Risk Assessment Modeling Approach and Data Sources

The analysis described in this section and in the Technical Background Document and appendixes is designed as a national analysis to assess the potential risk attributable to inhalation exposures when certain chemicals and metals are managed as a waste in certain types of waste management units. Of particular interest are chemicals and metals managed as wastes that are not regulated under RCRA as hazardous wastes. The purpose of this analysis is to determine which chemicals and waste management units are of potential national concern purely from a risk perspective; it is not intended to draw conclusions concerning regulatory coverage. This information, combined with preliminary information presented in the May 1998 Air Characteristic Study on regulatory coverage and on the presence of these chemicals in nonhazardous waste, will be useful in determining the possible need to expand regulatory coverage in the future.

The Air Characteristic Study addresses:

- # 105 constituents
- # 4 WMU types
  - landfill
  - land application unit (LAU)
  - wastepile (WP)
  - tank
- # 5 receptors
  - adult resident, exposure starting age 19 years
  - child resident, exposure starting age 0-3 years
  - child resident, exposure starting age 4-10 years
  - child resident, exposure starting age 11-18 years
  - off-site worker
- # direct inhalation only
- # volatiles and particulates
- # 6 distances from the site
- # 3 risk endpoints or averaging times
  - chronic (over 1 year)
  - subchronic (1 month) - LAU, WP
  - acute (1 day) - LAU, WP

This section provides a general overview of the approach and primary data sources used and discusses the major components of the analysis–emissions modeling, dispersion modeling, and exposure modeling/risk estimation. Technical details on the models and a complete set of inputs and associated references are provided in Volume II.

3.1 Overview of Modeling Approach

The overall goal of this risk analysis is to estimate the concentrations of constituents that can be present in a waste management unit (WMU) and remain protective of human health. These protective waste concentrations were calculated for 104 constituents[1] including volatiles, semi-volatiles, and metals. These

[1] 105 were addressed but one constituent, 3,4 dimethylphenol, did not have an inhalation benchmark.

constituents were selected for their potential to result in risk from inhalation exposure. Workers, adults, and children were evaluated for three different types of exposures or risk endpoints: chronic (over 1 year), subchronic (1 month), and acute (1 day). Estimating protective concentrations required a multistep modeling process that could relate the concentrations in ambient air at a receptor point that could create a health effect to a concentration in the waste management unit. To achieve this, the analytical approach for this analysis is based on three primary components:

- # Emissions modeling—characterizing emissions from a WMU
- # Dispersion modeling—describing the transport of these emissions through the ambient environment
- # Exposure modeling/risk estimation—estimating exposure to a receptor and then backcalculating to arrive at a waste concentration (C_w) that presents a risk equal to a prespecified risk level (e.g., 1 in 1 million, or 1E-6).

To illustrate the scenario that was modeled for this study, Figure 3-1 is a conceptual diagram of a waste site. Constituents managed in the WMU can be released as gases if they volatilize and as particulates if the constituent attaches to solid particles in the waste. Once the constituent is released from the site, the ambient air provides a medium for the transport of the airborne constituent. The direction the constituent travels and its concentration in the air are determined by meteorological conditions in the surrounding area such as wind direction, air temperature, and atmospheric stability at the time it is released. Because meteorological patterns are dynamic, the concentration of the constituents in the air varies over time and people who live and work at various locations around the WMU have different inhalation risks. The risk to an individual from the release of a constituent also depends upon characteristics of that individual such as body weight, inhalation rate, and the length of time that individual remains in the area around the WMU. These last characteristics are the reason that this assessment considers the

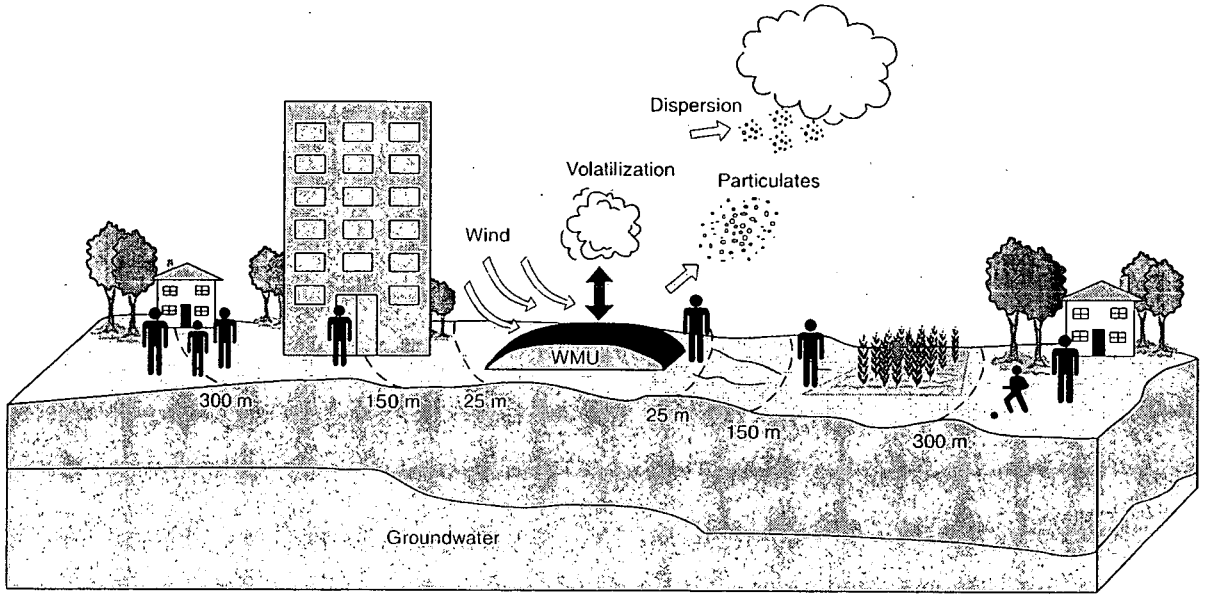


Figure 3-1. Conceptual diagram of a waste site.

31

exposure to multiple types of receptors: adult residents, child residents of various ages, and workers.

In order to model the scenario described above, the preliminary requirements for the analysis included:

- # Emissions models for the various WMUs to provide estimates of gas and particle releases from the unit
- # A dispersion model capable of modeling area sources for chronic (over 1 year), subchronic (1 month), and acute (1 day) releases
- # An exposure model for locating receptors proximate to the WMUs and estimating their exposure
- # A risk model that combines the exposure characteristics of different types of receptors with constituent-specific toxicity benchmarks.
- # The ability to backcalculate C_w from a prespecified risk level (e.g., 1E-6).

For each constituent and each WMU type, EPA wanted to be able to specify a C_w that would not exceed a target risk level (e.g., 1 in 100,000, or 1E-5) in more than a specified percentage (e.g., 10 percent) of the cases being modeled. Therefore, a probabilistic modeling approach, which would produce a distribution of C_w's, was needed, as opposed to a deterministic approach, which would only produce a point estimate. A deterministic analysis produces a point estimate because it uses a single value for each parameter in the analysis. A probabilistic approach considers the variability in the inputs required to estimate the concentration nationally. This type of approach produces a distribution of results because the method iterates through the analysis more than once, allowing the input parameters in the analysis to take on different values for each iteration from a distribution of values. For this analysis, EPA used a Monte Carlo simulation. This is a type of probabilistic analysis that can be used when the distribution of some or all input variables is known or can be estimated. A large number of iterations of the calculations are performed (i.e., 1,000), with a value for each input variable selected at random from the variable's distribution and the result (in this case, C_w) calculated for each iteration. The results of each iteration are combined into a distribution of C_w. It was assumed that the modeled cases represent the national distribution of risk-specific concentrations.

The probabilistic approach described above was used to model chronic exposures. A deterministic approach designed to produce a more high-end point estimate was used to model acute and subchronic exposures. The acute/subchronic approach uses the maximum exposure point at any given distance; so no variability in receptor location is accounted for. It also uses the meteorological conditions that produce the maximum air concentration for a 24-hour or 30-day time period over 5 years of meteorological data. The results from the acute/subchronic analysis are comparable to the 100th percentile of the distribution generated for the chronic analysis. It should be noted that acute/subchronic exposures were only assessed for land application units (LAUs) and wastepiles, which may have episodic loading events. There are a variety of other

38

differences in the acute/subchronic approach in how the emission rates and dispersion factors were calculated; these are described in more detail in Sections 3.2.2 and 3.2.3.

To estimate volatile emissions from each type of WMU, EPA's CHEMDAT8 model was used. For the landfill, LAU, and wastepile, the concentration of hazardous constituent in the surface layer of the soil (hereafter referred to as soil concentration) was estimated using a mass balance approach (i.e., competing pathways such as volatilization, adsorption, and biodegradation are accounted for). Particulate emissions due to wind erosion were modeled for land-based units (landfills, LAUs, and wastepiles). Landfills and LAUs were modeled as ground-level sources using the Cowherd model (U.S. EPA, 1985b, 1988). Wastepiles were modeled as elevated sources using the AP-42 model for wind erosion from aggregate storage piles (U.S. EPA, 1985a). To obtain the emission rate of constituent sorbed to particulate matter, the emission rate of particulate matter was multiplied by the soil concentration calculated by CHEMDAT8. This was done to account for the portion of the original constituent concentration that would remain in the waste after volatilization and biodegradation losses, and so would realistically be available for emission in the particulate phase.

The modeling assumes waste is continuously added to landfills and tanks, while LAUs and wastepiles have noncontinuous, episodic waste loadings. To capture potential peaks in emissions immediately after episodic loading events, acute and subchronic exposures were evaluated for LAUs and wastepiles.

Dispersion modeling was performed for each WMU using EPA's Industrial Source Complex Model Short-Term (ISCST3) to develop unitized air concentrations (UACs) for vapors and particulates. UACs are dispersion coefficients based on a unit emission (i.e., 1 $\mu g/m^2 \cdot s$) for use in a backcalculation. UACs varied depending on the averaging time (i.e., chronic, subchronic, or acute), the size of the WMU, the distance and direction of the receptor from the WMU, and the associated meteorological station. Dispersion modeling for vapors did not account for depletion, as sensitivity analysis showed that depletion of vapors has a negligible impact on air concentration of vapors. Dispersion modeling for particulates accounted for dry depletion of particles, since a sensitivity analysis showed that dry depletion has a potentially significant impact on air concentrations of particulates. Wet depletion of particulates was not accounted for in the dispersion modeling, as sensitivity analysis showed that wet depletion has little impact on air concentration.

The air concentration at any specific receptor is the product of the emission rate (in $\mu g/m^2 \cdot s$) and appropriate UAC (in $[\mu g/m3]/[\mu g/m^2 \cdot s]$). Air concentrations were estimated for chronic, subchronic, and acute exposures (using averaging times of 1 year, 1 month, or 1 day), based on a combination of volatile and particulate emissions.

Many previous risk analyses have used the maximum point of exposure at some prespecified distance from the WMU as the point for analysis. Such an approach is usually criticized as being overly conservative because it does not consider the possibility of no one living at that exact point. Because individuals may potentially be located in any direction and at various distances from a facility, this analysis developed an explicit way to incorporate this consideration. First, a sensitivity analysis was conducted to determine a reasonable distance at which to bound the analysis. This sensitivity analysis showed that, beyond 1,000 m, most air

concentrations are a small percentage (less than 10 percent) of the concentration at the point of maximum exposure. Therefore, 1,000 m was used as the outer bound on the distance of receptors included in this analysis. A receptor grid was set up to allow individuals to reside in any of 16 directions and at distances of 25, 50, 75, 150, 500, and 1,000 m from the edge of the unit.

For this analysis, five receptors were included: an adult resident, a child resident with exposure starting between 0 and 3 years old, a child resident with exposure starting between 4 and 10 years old, a child resident with exposure starting between 11 and 18 years old, and an off-site worker. These receptors could be located in any of 16 directions and at distances of 25, 50, 75, 150, 500, and 1,000 m from the edge of the unit. Each distance was evaluated separately and the location of a receptor was allowed to vary among any of the 16 directions. The 16 directions were equally weighted, so there is equal probability of a receptor's being located anywhere around the WMU. For acute and subchronic exposures, receptors were modeled at 25, 50, and 75 m because it was assumed that the greatest possibility of acute exposure would be closest to the site.

3.2 Conducting the Analysis

As discussed earlier, the analysis consists of three main parts: emissions modeling, dispersion modeling, and exposure modeling/risk estimation. Figure 3-2 shows the model framework. Emissions and dispersion modeling was performed first and the results used as inputs to the exposure modeling/risk estimation. In addition, a database containing characterizations of WMUs was used. The goal of the analysis is to backcalculate a waste concentration that will result in a specified risk. Because risk is assumed to be linear with waste concentration under most circumstances, a waste concentration was generated by forward-calculating a risk associated with a unit concentration in the waste (i.e., 1 mg/kg for land-based units and 1 mg/L for tanks), then scaling the unit concentration using the ratio of target risk to calculated risk. The assumption of linearity is accurate for the dispersion modeling and the exposure and risk modeling. The emissions model is linear for land-based units and tanks without biodegradation. The emissions model for tanks with biodegradation is nonlinear at the concentration where biodegradation shifts from first order to zero order. The results for tanks with biodegradation were backcalculated using first-order emission rates; however, if this result exceeded the concentration at which biodegradation becomes zero order, the result was adjusted to be based on zero-order emission rates. Even when the emissions model is linear, it is possible, using this approach, to backcalculate waste concentrations that exceed the solubility or soil saturation concentration for the chemical. Results that exceed the solubility or soil saturation concentration under neutral conditions are footnoted in the result tables (soil saturation concentration and solubility can vary according to site-specific temperature and pH conditions).

Emissions modeling was performed for all WMUs and all chemicals, assuming a unit concentration of the chemical in the waste (1 mg/kg for land-based units or 1 mg/L for tanks). These emissions were used as inputs to Step 2 of the exposure modeling/risk estimation portion of the model.

Dispersion modeling was performed for 76 representative WMU areas and height combinations and 29 meteorological locations, assuming a unit emission rate of 1 $\mu g/m^2$-s. This

34

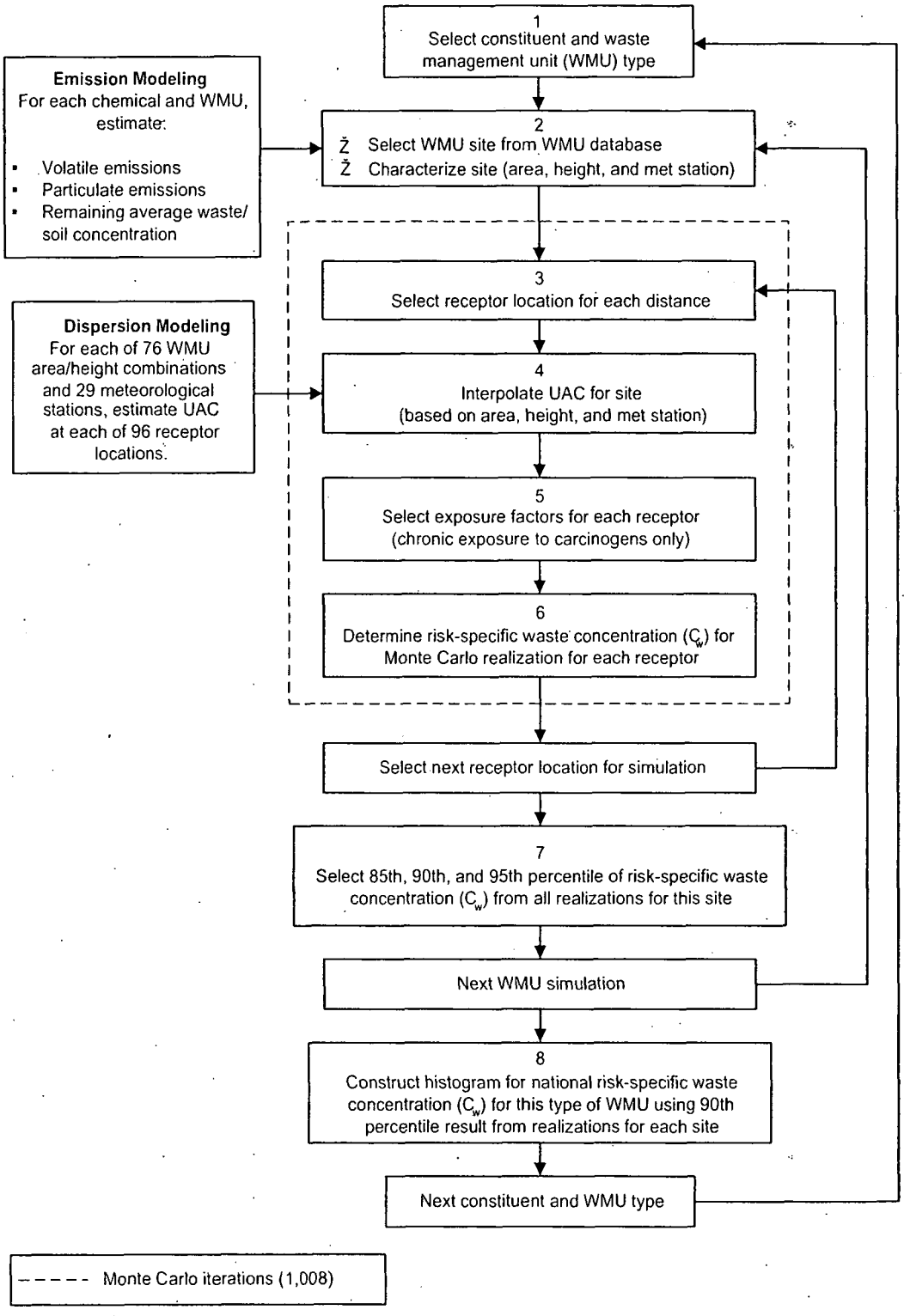


Figure 3-2. Model framework.

35

produced vapor and particle-phase UACs for each area/height combination, meteorological station, and receptor location, which were used as the basis from which to interpolate in Step 4 of the exposure modeling/risk estimation portion of the model.

The analytical framework shown in Figure 3-2 consists of a series of steps and loops. In Step 1, a chemical and WMU type (e.g., landfills) were selected (thus, all landfills were analyzed as a group for each chemical, and so on).

In Step 2, a WMU was selected from the data file for that unit type. For example, for landfills, the database has a data record containing the facility identification and WMU characteristics such as surface area, depth, and waste quantity managed per year for each of 801 landfill units. The database also has a sampling weight for each facility that defines how many facilities nationally were represented by that facility. An assigned meteorological station was added to the database based on locational information for each WMU. The model simulation starts with the first record and moves to each successive record. For each WMU record, the associated emission rate for that WMU and chemical was obtained from the emission modeling results.

In Step 3, receptor locations were selected by choosing at random one of the 16 directions modeled in the dispersion modeling. Receptors were modeled in that direction at each of six distances from the site.

In Step 4, a UAC was interpolated for the WMU. Due to the long run time of ISCST3 for area sources, UACs were modeled for only 76 selected WMU area/height combinations for each meteorological station and receptor location. To calculate a UAC corresponding to the WMU's actual area and height, EPA first chose the modeled height closest to the actual unit height, then interpolated between the UACs for the two closest of the areas modeled. For example, the first three areas modeled for wastepiles were 20, 162, and 486 m^2. These were modeled at heights of 1, 2, 4, 6, and 8 m. For a WMU with an actual area of 100 m^2 and an actual height of 3.5 m, the UAC was interpolated from the UACs for 20 m^2/4 m high and 162 m^2/4 m high. For a WMU with an actual area of 200 m^2 and an actual height of 6.9 m, the UAC was interpolated from the UACs for 162 m^2/6 m high and 486 m^2/6 m high.

In Step 5, for chronic exposures to carcinogens, values of exposure factors such as body weight, inhalation rate, and exposure duration were chosen at random from distributions of these parameters (developed from data in the *Exposure Factors Handbook*, U.S. EPA, 1997c and 1997d) to capture the variability in exposure factors for a given receptor. These exposure factors differ for different receptor types (such as adults, children, and workers). Noncarcinogens were not assessed in this manner because the health benchmarks, such as EPA's reference concentration (RfC), are expressed in terms of ambient concentration and cannot be adjusted for variations in these exposure factors. Similarly, acute and subchronic health benchmarks are expressed as ambient exposure concentrations and cannot be adjusted for variability in exposure factors.

In Step 6, the emission rate, UACs, and, if applicable, the exposure factors, were combined with the health benchmark for the chemical to estimate risk (for chronic exposure to carcinogens) or hazard quotient (for acute and subchronic exposures, and chronic exposures to

36

noncarcinogens) associated with the unit concentration modeled. This risk was then compared to the target risk of 1 in 1 million, 1 in 100,000, or 1 in 10,000 (i.e., 1E-6, 1E-5, or 1E-4) for carcinogens, and the ratio was used to scale the unit concentration to a concentration in the waste (C_w) that would result in the target risk at that receptor. A similar technique was used for scaling the hazard quotient for noncarcinogens.

Steps 3 through 6, which form the core of the Monte Carlo simulation, were then repeated 1,008 times for each WMU, resulting in a distribution of C_w for that WMU for each receptor (adult, child, or worker) at each distance from the site (25, 50, 75, 150, 500, and 1,000 m) for a specific risk criteria (i.e., 1E-4, 1E-5, or 1E-6 for carcinogens and 10, 1, or 0.25 for noncarcinogens). Once 1,008 iterations had been performed for a WMU, various percentiles were selected from the distribution to characterize it. These percentiles represent the percentage of receptors protected at the WMU.

Steps 2 through 6 were then repeated to obtain distributions of C_w for each WMU in the database. These distributions are somewhat different for carcinogens and noncarcinogens and for chronic, subchronic, and acute exposures. For chronic exposure to carcinogens, they represent both the potential variability in location around a WMU, as well as the variability in exposure duration, inhalation rate, and body weight for each receptor type. For noncarcinogens and for subchronic and acute exposures, variability in these exposure factors is not considered because the measure of risk is a ratio of air concentrations. For chronic exposures to noncarcinogens, the distributions represent the variability in location around the WMU at a specific distance. For subchronic and acute exposures, only point estimates were made at various distances using the receptor located at the point of maximum air concentration for that distance.

The cumulative distribution of C_w for each WMU is presented as the percentage of receptors that are at or below the risk criteria for any C_w (see Figure 3-3, left side). For example, 90 percent of all adult residents at a distance of 150 meters have a predicted risk at or below 1 in 100,000 (1E-5) if the concentration of the chemical (e.g., cumene) in the landfill is 1 mg/kg (see point a). A second landfill may have a 90 percent protection level for all adult residents at 150 m at a concentration of 10 mg/kg (point b), and a third landfill at a concentration of 100 mg/kg (point c). Thus, in Step 7, for each WMU, the distribution shows the percent of potential receptors at or below a specified risk level for each concentration of constituent in the WMU (C_w) for each distance and each receptor.

In Step 8, once all WMUs of a certain type had been modeled, the distributions of C_w for all individual WMUs of the same type (e.g., landfills) were combined to produce a cumulative distribution that presents the variability in C_w across all units of a certain type. For a given percentage of protected receptors (e.g., 90 percent) as described above, the C_w was combined across all WMUs of a specified type (e.g., landfills) to provide a distribution of the percentage of sites considered protective at that level, as shown in Figure 3-3 (right side). Figure 3-3, for example, shows the cumulative distribution of C_w at a 90 percent protection level across all landfills. From this distribution, the 90th percentile C_w value for all 90 percent protection levels across all landfills could be estimated. As described above, three landfills that give a 90 percent protection level (i.e., at 1E-5) for a resident at 150 meters from the unit boundary have corresponding C_w values of 1 mg/kg, 10 mg/kg, and 100 mg/kg (see points labeled a, b, and c).

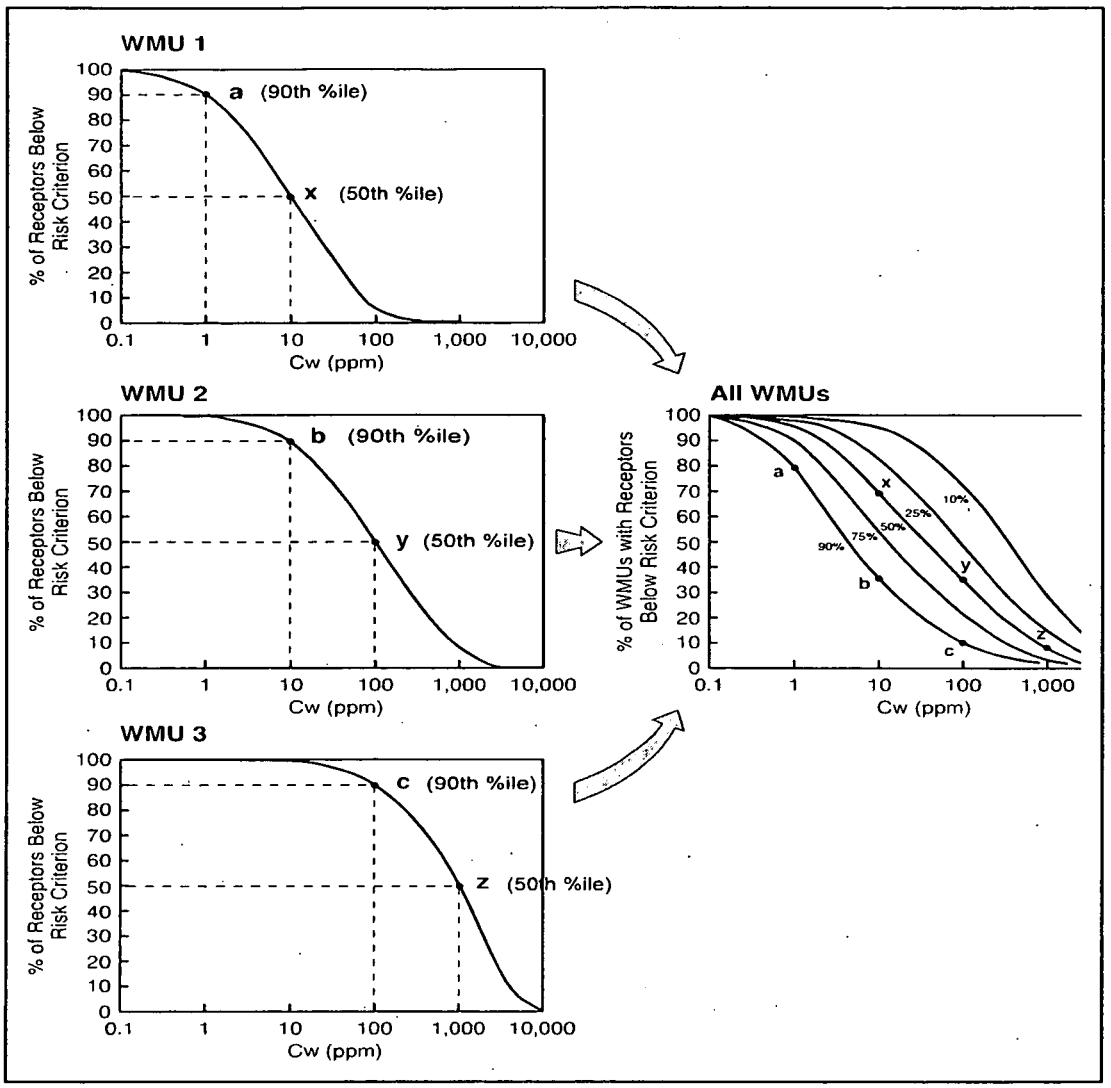


Figure 3-3. Combination of results for individual WMUs into a distribution across all WMUs.

These values plus similar values from all other landfills constitute the cumulative distribution. The C_w value that is protective of 90 percent of receptors across 90 percent of the sites is referred to in this study as the 90/90 protection level. These distributions were developed for each unit type, each receptor type, each risk criteria, and each distance from the WMU.

These cumulative distributions are intended to encompass the variability across WMUs. Thus, the variability in WMU characteristics and in meteorological settings is included in these distributions.

This process was repeated from Step 1 for each chemical and WMU type analyzed in this study.

38

3.2.1 Data Sources

The Industrial D Survey database (Shroeder et al., 1987) was the primary source of data on WMUs used in this analysis. This database provides information on each of the WMUs assessed, with the exception of tanks. Tank data are from EPA's National Survey of Hazaradous Waste Treatment, Storage, Disposal and Recycling Facilities (TSDR Survey, U.S. EPA, 1987). The Industrial D Survey database contains information on the size and capacity of a statistical sample of each WMU type, general location information, and statistical weights for each facility in the sample. The statistical sample was designed to represent all industrial waste management units not regulated under the RCRA hazardous waste program at the time the survey was conducted in 1987. The weights in the database indicate the number of facilities represented by each facility in the sample. For this assessment, it is assumed that the data contained in this database provide an appropriate representation of the characteristics of each WMU type and of the general location of these types of facilities with respect to climate regions of the country.

Meteorological stations provided temperature and windspeed data as inputs to the emissions model and a large set of inputs for the dispersion model. Although meteorological data are available at over 200 meteorological stations in the United States (see, for example, *Support Center for Regulatory Air Models (SCRAM) Bulletin Board* at http://www.epa.gov/scram001), various resource constraints prevented the use of all available data sets in this analysis. Therefore, a set of 29 stations was used that had been selected as representative of the nine general climate regions in the contiguous United States in an assessment for EPA's Superfund Soil Screening Level (SSL) program (EQM, 1993).

In EPA's Superfund study, it was determined that 29 meteorological stations would be a sufficient sample to represent the population of 200 meteorological stations and predict mean dispersion values with a high (95 percent) degree of confidence. The 29 meteorological stations were distributed among the nine climate regions based on meteorological representativeness and variability across each region. Large-scale regional average conditions were used to select the actual stations.

The 29 meteorological stations are listed in Section 5 of Volume II. To assign each Industrial D or TSDR facility to a meteorological station, EPA used a geographic information system (GIS) to construct areas around each station that encompass the areas closest to each station. The boundaries of these areas were then adjusted to ensure that each boundary encloses an area that is most similar in meteorological conditions to those measured at the meteorological station. First, the boundaries were adjusted to correspond to Bailey's ecological divisions (Bailey et al., 1994), which are defined primarily on physiography and climate. The boundaries were further adjusted for coastal (including Great Lakes) areas and the central valley of California to ensure that these stations were used only in regions with similar meteorology. Based on zip codes in the Industrial D Survey database and EPA IDs in the TSDR database, the sites were then overlaid on this GIS coverage, and meteorological station assignments were then exported for use in the modeling exercise. Several sites in Alaska, Hawaii, and Puerto Rico were deleted from the analysis at this point because the 29 meteorological stations are limited to the continental United States.

Figure 3-4 shows the final meteorological station boundaries used for the study along with the zip code centroid locations for the Industrial D sites.

3.2.2 Emissions Modeling

Both volatile emissions (for all WMU types) and particulate emissions due to wind erosion (for land-based WMUs) were included in the risk analysis. To assess these two types of emissions, three parameters had to be modeled: volatile emission rate, long-term average soil concentration in the unit (for LAUs, landfills, and wastepiles), and particulate matter emission rate.

EPA's CHEMDAT8 model was selected as the model to estimate volatile emissions rates and long-term average soil concentrations in the WMU. The CHEMDAT8 model was originally developed in projects funded by EPA's Office of Research and Development (ORD) and Office of Air Quality Planning and Standards (OAQPS) to support National Emission Standards for Hazardous Air Pollutants (NESHAP) from sources such as tanks, surface impoundments, landfills, wastepiles, and land application units for a variety of industry categories including chemical manufacturers, pulp and paper manufacturing, and petroleum refining. It also has been used to support the emissions standards for hazardous waste treatment, storage, and disposal facilities (TSDF) (U.S. EPA, 1991) regulated under Subpart CC rules of RCRA, as amended in 1984. The CHEMDAT8 model is publicly available and has undergone extensive review by both EPA and stakeholder representatives. The CHEMDAT8 spreadsheet model and model documentation may be downloaded at no charge from EPA's web page (http://www.epa.gov/ttn/chief/software.html).

The CHEMDAT8 model considers most of the competing removal pathways that might limit air emissions, including adsorption, hydrolysis (for tanks only), and biodegradation. Adsorption/absorption is the tendency of a chemical or liquid media to attach or bind to the surface or fill the pores of particles in the soil or waste and therefore not volatilize into the air. This tendency to adsorb to or absorb in particles is an important process for estimating the concentration of the chemical on particles emitted to the air due to wind erosion. CHEMDAT8 in its original form models adsorption for land-based units by presuming that the entered waste concentration is in liquid phase. Because waste concentrations are more typically measured as total concentration (liquid plus solid phase), CHEMDAT8 was modified to model adsorption explicitly for an entered total waste concentration for land-based units. Biodegradation is the tendency of a chemical to be broken down or decomposed into less complex chemicals by organisms in the waste or soil. Similarly, hydrolysis is the tendency of a chemical to be broken down or decomposed into less complex chemicals by reaction with water. Chemicals that decompose due to biodegradation or hydrolysis have lower potential for emission to the air as gases or particles than those that do not. Loss of contaminant by leaching or runoff is not included in the CHEMDAT8 model. Both leaching and runoff are a function of a chemical's tendency to become soluble in water and follow the flow of water (e.g., due to rainfall) down through the soil to groundwater (leaching) or downhill to surface water (runoff). These two mechanisms would also make less chemical available for emission to the air as a gas or as particles. As such, CHEMDAT8 is considered to provide reasonable to slightly high (environmentally protective) estimates of air emissions from the land-based units.

40

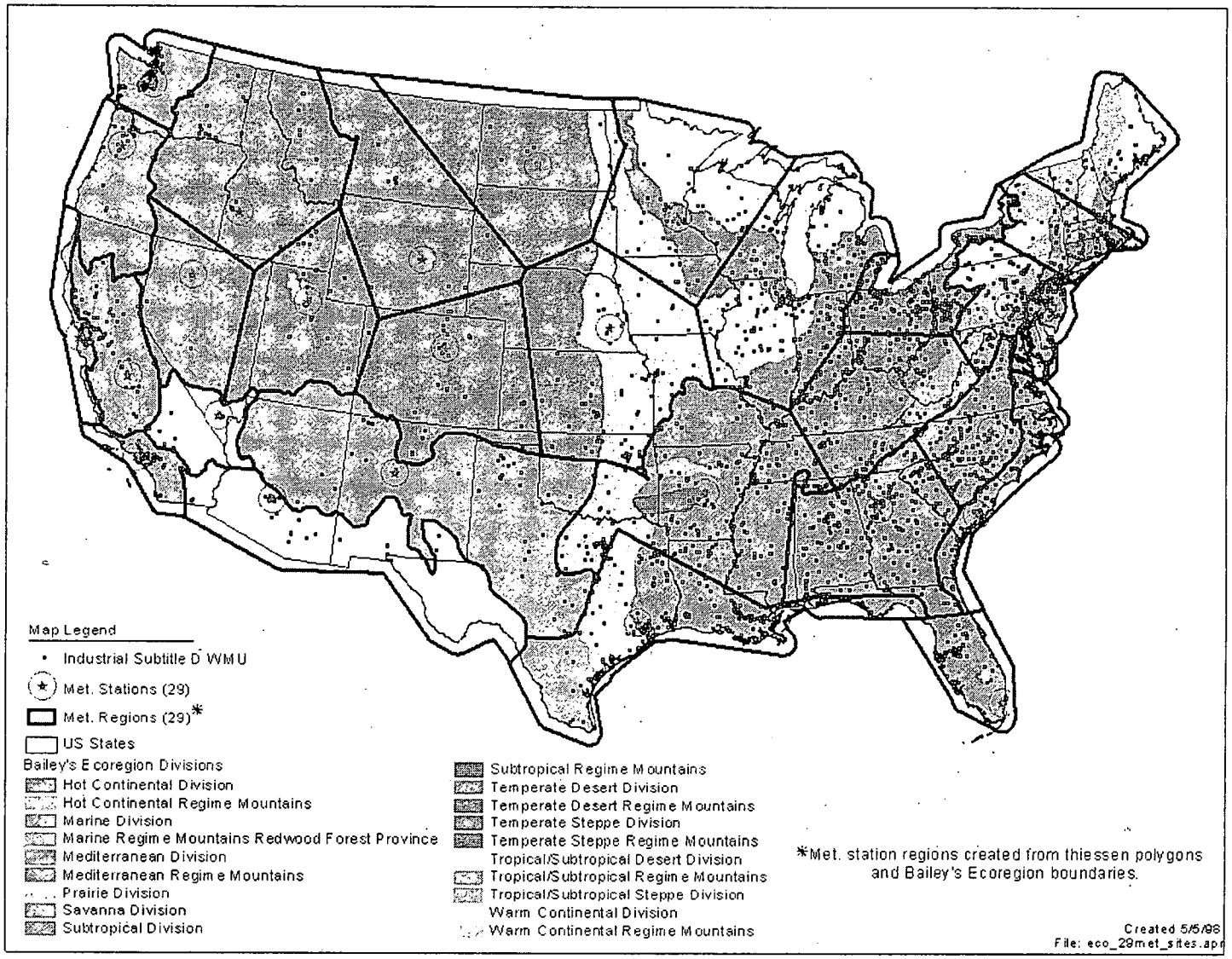


Figure 3-4. Meteorological station regions.

41

Two different models were used to model wind erosion: one for wastepiles (elevated sources) and one for landfills and land application units (ground-level sources). The Cowherd model (U.S. EPA, 1985b and 1988) was selected for modeling wind erosion emissions from ground-level sources, and the AP-42 model for wind erosion from aggregate storage piles (U.S. EPA, 1985a) was selected for modeling wind erosion emissions from wastepiles. Newer versions of both of these models are available; however, the newer versions are event-based algorithms that require extensive site-specific data that were not available for the sites modeled in this analysis. The versions used probably result in somewhat higher particulate emissions estimates than the event-based algorithms would. This overestimation of particulate emissions is not significant for volatile chemicals, as particulate emissions were found to be a negligible fraction (less than 2 percent in most cases) of total emissions for the volatile chemicals modeled in land-based units. The protective waste concentrations (C_w's) for metals other than mercury (which do not volatilize and are therefore based solely on particulate emissions) may be somewhat lower as a result of this overestimation of emissions.

Both volatile and particulate emissions were estimated for the landfill, land application unit, and wastepile, while only volatile emissions were estimated for tanks.

3.2.2.1 Estimating Volatile Emissions. The modeling scenario and critical parameters required for each type of WMU are provided in the following subsections. A more detailed discussion of the emissions modeling is provided in Volume II.

The input parameters used for the CHEMDAT8 land-based unit emissions model are presented in Table 3-1.[2] Of these parameters, two are actually flags to determine which model equations to apply: the aqueous waste flag and the biodegradation flag. The most important flag for emission estimates is probably the aqueous waste flag. This flag tells the CHEMDAT8 model which equilibrium partitioning model to use between the liquid and gas phases. For organic wastes, the model uses Raoult's law and the liquid-to-air partition coefficient becomes proportional to the contaminant's partial vapor pressure. For aqueous wastes, the model uses Henry's law and the liquid-to-air partition coefficient becomes proportional to the contaminant's Henry's law coefficient.

All land-based WMUs were modeled twice; once assuming unit concentration (concentration set to 1 mg/kg, assuming Henry's law applies) and once assuming pure component (concentration set to 1E+6 mg/kg, assuming Raoult's law applies). The results presented in Section 4 and in Volume III are based on the aqueous phase emission rates (unit concentration and Henry's law). The pure component emission rates were used only to identify chemicals for which greater emissions occur from the organic phase than from the aqueous phase (which is rare) or to identify chemicals for which the aqueous-based results exceeded soil saturation concentrations or solubility limits at neutral pH and standard temperature and to note for these whether the target risk or hazard quotient would be exceeded modeling pure component.

[2]The data entry form in the CHEMDAT8 model refers to oil rather than waste; the term waste is used here for clarity.

42

Table 3-1. CHEMDAT8 Land-Based Unit Model Input Requirements

Input Parameter	Data Source/Assumption
Loading (g waste/cm^3 soil)	Waste quantity and/or density from Ind. D Survey
Concentration in waste (ppmw)	1 for unit concentration run; 1E+6 for pure component run
Depth of tilling (or unit) (cm)	Assumed or set by capacity
Total porosity	Assumed default value of 0.5
Air porosity (0 if unknown)	Assumed default value of 0.25
Molecular weight of waste (g/mol)	18 for unit concentration run; 147 for pure component run
Aqueous waste flag:	For aqueous waste, enter 1 For organic waste, enter 0
Time of calculation (days)	Dependent on type of WMU
Biodegradation Flag:	For biodegradation, enter 1 For no biodegradation, enter 0
Temperature (°C)	Set by location of WMU
Windspeed (m/s)	Set by location of WMU
Area (m^2)	Input from Ind. D Survey
Fraction organic carbon	Assigned randomly from distribution

Three other parameters are critical for land-based units: the annual waste quantity, the temperature, and the biodegradation rate. The annual waste quantity, along with assumptions regarding the frequency of waste addition and the dimensions of the WMU, combine to influence a number of model input parameters including loading, concentration of contaminant in the waste, depth of the unit (or tilling), operational life, and surface area of the WMU.

Temperature is important because it affects the air diffusivity, which affects the volatilization rate and may affect the biodegradation rate (biodegradation rates were independent of temperature above 5°C and were set to zero below 5°C). Temperature is the only meteorological data input that potentially impacts the emissions results for the CHEMDAT8 model for the land-based WMU. The CHEMDAT8 model is insensitive to windspeeds for long-term emission estimates from land-based units.

The process of biodegradation is important because it lowers both the emission rate and the average soil concentration. Consequently, biodegradation is an important input parameter, and the biodegradation rate constants used in the model are critical parameters. Biodegradation was treated differently for the various WMUs. Landfills are not designed for biodegradation, and waste in wastepiles managed over short periods will not be affected substantially. Therefore, both the landfill emission runs and the short-term wastepile emission runs did not include biodegradation losses. First-order biodegradation was included in the LAU emission runs and long-term wastepile emission runs.

43

Table 3-2 presents the required CHEMDAT8 input parameters for tanks. Three types of parameters are critical: factors affecting turbulence, waste characteristics affecting biodegradation, and meteorological inputs.

Factors that affect the relative surface area of turbulence and the intensity of that turbulence are important in determining the fate of chemicals in tanks. The tank model has several input parameters that impact the degree and intensity of the turbulence created by the aeration (or mixing). The tank model is most sensitive to the fraction aerated.

Waste characteristics that influence the rate of biodegradation are important in determining emissions from both aerated and storage tanks. As shown in Table 3-2, these parameters include active biomass concentration, total solids in, total organics in, and total biorate. Biodegradation was modeled for aerated tanks reporting biological treatment. Aerated tanks reporting other types of treatment, nonaerated treatment tanks, and storage tanks were modeled with no biodegradation.

Unlike the biodegradation rate model that was used for the land-based units, the biodegradation rate model used in CHEMDAT8 for tanks depends on the amount of active biomass in the WMU. Therefore, the active biomass concentration is a critical parameter for

Table 3-2. CHEMDAT8 Tank Model Input Requirements

Input Parameter	Date Source/Assumption
Unit Design	
Flow rate (m^3/s)	Survey
Depth (m)	Imputed based on volume
Average surface area (m^2)	Imputed based on volume and depth
Height above ground (m)	Imputed based on depth
Aeration Parameters	
Fraction agitated	Estimated distribution
Total power (hp)	Imputed based on volume
Number of impellers	Imputed based on total power
Impeller diameter (cm)	Estimated constant = 61
Impeller speed (rad/s)	Estimated constant = 130
Power efficiency (unitless)	Estimated constant = 0.83
O_2 transfer rate (lbO^2/h-HP)	Estimated constant = 3
Submerged air flow (m^3/s)	Estimated constant = 0
Waste Characteristics	
Active biomass conc. (kg/m^3)	Estimated distribution, depends on treatment code
Total solids in (kg/m^3)	Estimated distribution
Total organics (COD) In (g/m^3)	Estimated distribution
Total biorate (mg/g-h)	Estimated constant = 19
Meteorological Data	
Temp (°C)	Imputed based on meteorological station
Windspeed (m/s)	Imputed based on meteorological station

44

aerated tanks. Because this parameter can vary widely for different types of tanks, biomass concentrations were set on a tank-by-tank basis for aerated tanks using process code information (WMU codes) from the TSDR Survey.

Meteorological inputs are also important for the tank emission model. For nonaerated treatment tanks and storage tanks, the emission estimates are impacted by both temperature and wind speed. Because the emissions for aerated tanks are predominantly driven by the turbulent area and associated mass transfer coefficients, the emissions from the aerated tanks are not strongly impacted by the wind speed. Aerated tank emissions are impacted by temperature. Annual average temperatures were used as input to the model based on tank locations.

The following sections describe the emissions assumptions used for determining volatile emissions for each WMU type.

Landfills. For landfills, annual average emissions were estimated from the active landfill cell assuming the active landfill cell could hold 1 year's worth of waste material. The emissions for the active cell were made assuming that the cell is instantaneously filled and that no waste cover is applied for the first year. Therefore, a full year's worth of waste was available for emissions to the air each year. Once the cell is covered at the end of a year, no additional emissions of gases or particles were modeled from that cell. Because landfills are not constructed for the purpose of biodegrading wastes, as are land application units or biologically active tanks, and because conditions are not controlled to foster biodegradation in landfills, biodegradation was not modeled in landfills.

Assumptions for Modeling Volatile Emissions from Landfills

- Landfill operates for 20 years filling 20 equal cells sequentially.
- The active cell is modeled as being instantaneously filled at time t=0 and remains open for 1 year.
- Emissions are calculated only for one cell for 1 year (after 1 year, cells are either depleted of the constituent or capped).
- Waste is homogeneous with an initial concentration of 1 mg/kg.
- The waste matrix may be aqueous (Henry's law partitioning applies) or organic (Raoult's law partitioning applies).
- Annual average temperature is used (determined by assigned meteorological station).
- Acute and subchronic exposures were not modeled.

The annual average emission rate and waste concentration for the active landfill cell were estimated using annual average meteorological data. A sensitivity analysis showed no difference in emissions estimates using seasonal meteorological data (less than 2 percent error for most chemicals). The annual average emission rates were used for chronic risk calculations. Acute and subchronic risks were not considered for landfills.

The average concentration of the waste in the landfill cell was estimated from the emission fraction by assuming first-order contaminant (concentration) disappearance. The details of this calculation are provided in Volume II, Section 4. The relationship between the emission rate and the waste concentration was needed to estimate a concentration in the WMU that corresponded to a specific risk or hazard quotient (HQ) for a receptor.

Land Application Units. For land application units, the volatile emissions were estimated assuming waste additions 24 times per year and first-order biodegradation for temperatures greater than 5°C. The emissions were estimated using monthly average meteorological data. LAU emissions are time-dependent (depending on how recently waste was added) but were modeled as pseudo–steady-state (i.e., steady-state emissions were modeled for a series of short intervals; these estimates were then averaged to produce a long-term emission rate). The average emission rate and associated waste/soil mixture concentrations were estimated for each bimonthly period (i.e., the time between applications). These computations were carried out for 40 years, and the average emission rate and soil/waste concentration for year 40 was used to estimate the long-term annual average emission rates and soil/waste concentrations for each contaminant. The 40-year time period is long enough to result in steady-state emissions for most chemicals and is longer than most of the exposure durations used in the analysis. The annual average emission rates were used for chronic risk calculations. For acute and subchronic risk calculations, emissions were calculated based on the first 24 hours (for acute) or the first 30 days (for subchronic) after waste was added. In the absence of biodegradation, higher temperatures would produce higher volatile emissions. However, when biodegradation is modeled, it slows to zero at temperatures below 5°C, thus increasing volatile emissions at low temperatures. Therefore, for both acute and subchronic exposures, emissions were modeled at the maximum monthly temperature and the minimum monthly temperature, and the one that produced higher emissions was used.

Assumptions for Modeling Volatile Emissions from LAUs

- # Waste application occurs 24 times per year.
- # Emissions are modeled as pseudo–steady-state.
- # Emissions in year 40 are used to estimate long-term emissions.
- # Waste is homogeneous with an initial concentration of 1 mg/kg.
- # The waste matrix may be aqueous (Henry's law partitioning applies) or organic (Raoult's law partitioning applies).
- # Monthly average temperature was used for chronic exposure (determined by assigned meteorological station).
- # Maximum or minimum average monthly temperature was used for acute and subchronic exposures (whichever gave higher emissions).
- # Biodegradation occurs at temperatures greater than 5°C.

Wastepiles. The wastepile was assumed to remain at a constant volume. Annual waste additions were therefore matched with a corresponding quantity of waste removed. The average residence time of the waste (based on the size of the wastepile and the annual waste quantity) was used to estimate the emission rate and waste concentration across the wastepile. Monthly average emission/waste concentration estimates were made using monthly meteorological data and first-order biodegradation for temperatures greater than 5°C. The resulting monthly average emissions and waste concentrations were then arithmetically averaged to estimate the long-term annual average emission rates and waste concentrations for each contaminant. The annual average emission rates were used for chronic risk calculations. For acute and subchronic risk calculations, emissions were calculated based on the first 24 hours (for acute) or the first 30 days

46

(for subchronic) after waste was added. Because wastepiles are not designed for biodegradation, there may be a lag time after waste is added before enough acclimated biomass accumulates for biodegradation to begin. This time is typically several months, longer than the 1-day or 30-day periods modeled for acute and subchronic exposures. Therefore, biodegradation was not considered for wastepiles for acute and subchronic exposures. In the absence of biodegradation, higher temperatures produce higher emissions. Therefore, for both acute and subchronic exposures, emissions were modeled at the maximum monthly temperature.

Tanks. For all tanks, the emissions were estimated assuming units were well-mixed and were operating at steady state. The tanks were assumed to have a constant influent and were assumed to operate at a constant temperature. Annual average temperatures and windspeeds were used to estimate the operating conditions for the tanks. A biodegradation rate model using Monod kinetics was used to estimate biodegradation rates for aerated treatment tanks expected to have biodegradation (based on process codes). Biodegradation was not modeled for nonaerated treatment tanks and storage tanks.

Due to the nonlinearity of the biodegradation rate model used in the tank emission estimates, direct backcalculation of an acceptable waste concentration may not be appropriate for some compounds. Unlike the emission

Assumptions for Modeling Volatile Emissions from Wastepiles

- # Wastepile operates with a fixed volume.
- # Waste is homogeneous with an initial concentration of 1 mg/kg.
- # The waste matrix may be aqueous (Henry's Law partitioning applies) or organic (Raoult's law partitioning applies).
- # No specific operating life was assumed for wastepiles. Residence time of waste in the pile was unit specific.
- # Monthly average temperature was used for chronic exposure (determined by assigned meteorological station).
- # Maximum monthly temperature was used for acute and subchronic exposures.
- # Biodegradation occurs at temperatures greater than 5°C for chronic exposures.
- # No biodegradation was assumed for acute and subchronic exposures.

Assumptions for Modeling Volatile Emissions from Tanks

- # Tanks operate at steady state.
- # Tank is well mixed.
- # Waste has an influent concentration of 1 mg/L.
- # The waste matrix may be aqueous (Henry's Law partitioning applies) or organic (Raoult's law partitioning applies).
- # Annual average temperature was used for chronic exposure (determined by assigned meteorological stations).
- # Operating life is not an explicit input; assumed to be long enough to reach steady state.
- # Biodegradation rate is first order with respect to biomass concentrations.
- # Biodegradation rate follows Monod kinetics with respect to contaminant concentrations.
- # Hydrolysis rate is first order with respect to contaminant concentrations.
- # Acute and subchronic exposures were not modeled.

47

results from the land-based units, the contaminant concentration used in the analysis may impact the predicted "normalized" emission rate (i.e., the emission rate in g/m^2-s per mg/L of contaminant). Therefore, the tanks with biodegradation were run at a low concentration (i.e., 0.001 mg/L) and at a high concentration (i.e., the constituent's solubility). The most appropriate backcalculated emission value was then selected based on the concentration range of the backcalculated values and the constituent's biodegradation characteristics (see Volume II, Section 7.9, for further details).

3.2.2.2 Development of Particulate Emissions. Particulate emissions due to wind erosion were modeled for land-based units (landfills, land application units, and wastepiles). Particulate emissions from truck movement and other activities at the WMUs were not modeled. These activities are likely to result in short bursts of particulate emissions and should be modeled using an event-based emissions model. Such models require more site-specific information than was available for the sites modeled in this analysis.

Landfills and LAUs were modeled differently than wastepiles because they are ground-level sources and wastepiles are elevated sources. For both types of WMU, the models described in this section and in Volume II predict the emission rate of particulate matter released from a site due to wind erosion. To obtain the emission rate of constituent sorbed to particulate matter, the emission rate of particulate matter must be multiplied by the soil or waste concentration.

Landfills and Land Application Units. Wind erosion emissions from landfills and LAUs were modeled using the Cowherd model (U.S. EPA, 1985b). A newer version of Cowherd's model is available in U.S. EPA (1988). However, the newer version is an event-based model that requires detailed site-specific information unavailable for this analysis. Therefore, it was not used. The older Cowherd model tends to slightly overestimate emissions relative to the event-based version. Although the degree to which it overestimates is not known, it is expected to be relatively small. Because particulate emissions are negligible compared to volatile emissions for the volatile chemicals modeled, this is only of concern for the metals (other than mercury), which are based only on particulate emissions.

Inputs and Intermediate Values Used for Wind Erosion from Landfills and LAUs

Symbol	Parameter	Units	Value	Source
V	Vegetative cover	fraction	0	Assumption
z_0	Roughness height	cm	1	U.S.EPA (1985b)
U^*	Threshold friction velocity	m/s	0.5	Assumed unlimited reservoir

The Cowherd model estimates the emission of respirable particles (i.e., PM_{10}) due to wind erosion from a ground-level surface with an unlimited reservoir of erodible particles. Surfaces are defined as having a limited or unlimited reservoir based on threshold friction velocity (U^*). Surfaces with a U^* greater than 0.5 m/s are considered limited; those with U^* less than 0.5 m/s are considered unlimited (U.S. EPA, 1988). Threshold friction velocity is a measure of the windspeed at the ground surface that would be required to remove particles from the surface. Examples of limited reservoirs include nonhomogeneous surfaces with stones, clumps of

vegetation, or other nonerodible elements or crusted surfaces. Further, wind erosion is considered unlikely to occur from surfaces with full vegetative cover.

A detailed explanation of inputs used in the calculation of particulate emissions is presented in Section 4 of Volume II including vegetative cover, roughness height, average annual windspeed, and threshold friction velocity.

Wastepiles. Wind erosion emissions from wastepiles were modeled using an equation from AP-42 (U.S. EPA, 1985a) for estimating emissions from wind erosion from active storage piles. The equation gives emissions of total suspended particulates (TSP). Typically, an equation-specific particle size multiplier is applied to reduce the emissions to a desired size category, in this case, PM_{10}. No particle size multipliers are given for this equation in AP-42; however, Cowherd (U.S. EPA, 1988) gives a particle size multiplier of 0.5 for use with this equation, and this was used.

Important input parameters for this calculation include silt content of waste (i.e., percent with small particle size), number of days with greater than 0.01 inches of rainfall, and percent of time that windspeed exceeds 5.4 m/s. Data on the silt content of the wastes being modeled were not available. A median silt content of 12 percent was used based on "miscellaneous fill material" from AP-42 (U.S. EPA, 1985a). The number of precipitation days and the frequency of windspeed greater than 5.4 m/s were location-specific; values were obtained from NOAA (1992) and are summarized in Section 4 of Volume II.

3.2.3 Dispersion Modeling

Dispersion modeling was used to estimate air concentrations to which the various human receptors were exposed. A dispersion model (ISCST3) was run to calculate air concentrations associated with a standardized unit emission rate (1 $\mu g/m^2$-s) to obtain a unitized air concentration (UAC), also called a dispersion factor, which is measured in $\mu g/m^3$ per $\mu g/m^2$-s. Total air concentration estimates are then developed by multiplying the constituent-specific emission rates derived from CHEMDAT8 by this dispersion factor.

Assumptions Made for Dispersion Modeling

- Dry depletion was activated in the dispersion modeling for particulates. Depletion was not considered for vapors.
- An area source was modeled for all WMUs.
- To minimize error due to site orientation, a square area source with sides parallel to X- and Y-axes was modeled.
- Modeling was conducted using a unit emission rate of 1 $\mu g/s\text{-}m^2$.
- Receptor points were placed on 25, 50, 75, 150, 500, and 1,000 m receptor squares starting from the edge of the source with 16 receptor points on each square.
- The rural option was used in the dispersion modeling since the types of WMUs being assessed are typically in nonurban areas.
- Flat terrain was assumed.

Running ISCST3 to develop a dispersion factor for each of the approximately 3,400 individual WMUs modeled in this study would have been very time consuming due to the run time of the area source algorithm in ISCST3. In addition, modeling for many different locations requires extensive preprocessing to generate the detailed meteorological data needed for each location modeled. Therefore, a database of

dispersion factors was developed by running ISCST3 for many separate scenarios designed to cover a broad range of unit characteristics, including:

- # both ground-level and elevated sources
- # 14 surface area sizes for landfills and land application units, 29 surface area–height combinations for waste piles, and 33 surface area-height combinations for tanks
- # 6 receptor distances from the unit (25, 50, 75, 150, 500, and 1,000 meters) placed in 16 directions in relation to the edge of the unit.

Based on the size and location of a specific unit, an appropriate dispersion factor was interpolated from the database of dispersion factors using the closest location and the two closest unit sizes.

In addition, WMUs were assigned to and dispersion modeling was performed for 29 meteorological stations. These were chosen from the more than 200 available to represent the nine general climate regions of the continental U.S.

Each UAC in the database is specific to one meteorological station, one area-height combination, one distance from the unit, and one direction from the unit.

3.2.3.1 Model Selection

A number of dispersion models are available through the EPA Support Center for Regulatory Air Models (SCRAM) Bulletin Board (http://www.epa.gov/scram001/). These dispersion models were developed for a variety of applications, and each has its own strengths and weaknesses. This analysis required a model with the capability to model (1) area sources; (2) ground-level and elevated sources; (3) off-site impacts; (4) vapors and particulates; and (5) annual, monthly, and daily averaging times.

ISCST3 (U.S. EPA, 1995) was selected for all aspects of this analysis because it met all the criteria. This model, however, requires considerable run time, which limited the number of meteorological stations included in this analysis.

3.2.3.2 Meteorological Stations. As stated in Section 3.2.1, due to the considerable run time of ISCST3 for area sources, a set of 29 meteorological stations selected in an assessment for EPA's Superfund Soil Screening Level (SSL) program (EQM, 1993) as being representative of the nine general climate regions of the continental United States was used.

The dispersion modeling was conducted using 5 years of representative meteorological data from each of the 29 meteorological stations. Five-year wind roses representing the frequency of wind directions and windspeeds for the 29 meteorological stations were analyzed. These show that the 29 meteorological stations represent a variety of wind patterns. The wind roses are presented in Appendix C of Volume II.

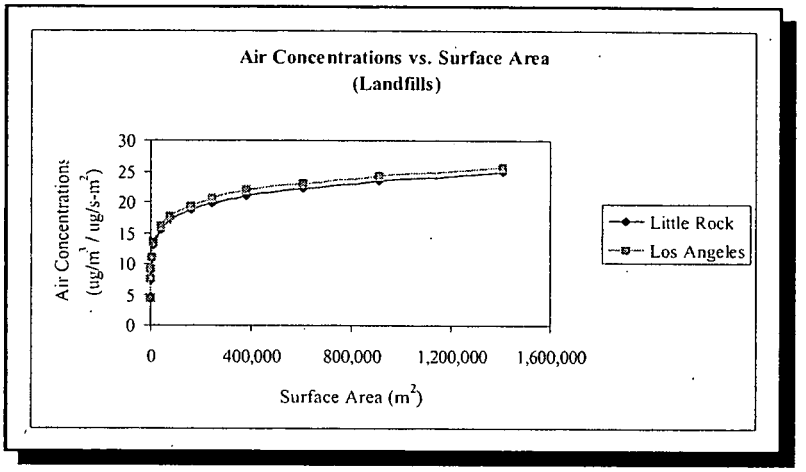
50

Wind direction and windspeed are typically the most important factors for dispersion modeling analysis. Wind direction determines the direction of the greatest impacts. Windspeed is inversely proportional to ground-level air concentrations, so that the lower the windspeed, the higher the air concentration. Mixing height and stability class are other meteorological conditions that influence dispersion. Mixing height determines the heights to which pollutants can be diffused vertically. Stability class is also an important factor in determining the rate of lateral and vertical diffusion. The more unstable the air, the greater the diffusion.

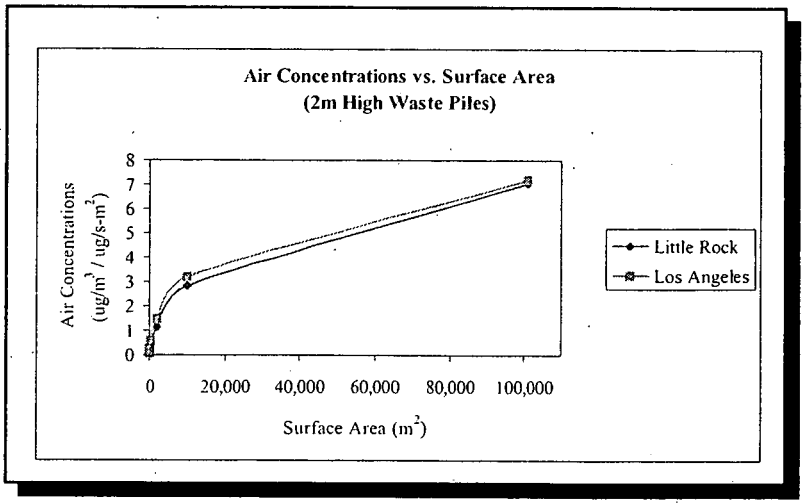
Shape of Wind Rose for 29 Meteorological Stations

Shape of Wind Rose	No. of Stations
Narrowly distributed	10
Moderately distributed	4
Evenly distributed	6
Bimodally distributed	9

3.2.3.3 Source Release Parameters. In the modeling analysis, four types of WMUs were considered (landfill, land application unit, wastepile, and tank). Because ISCST3 is sensitive to the size of the area source, the relationship between air concentrations and size of the area source was analyzed. The results show that, for relatively small area sources, air concentrations increase significantly as the size of the area source increases (see boxes). For large area sources, this increase in air concentrations is not as significant. To address this model sensitivity yet avoid modeling approximately 3,400 separate WMUs, EPA developed area strata that represented the distribution of the surface area for each of the WMU types. The surface areas were then used in the dispersion modeling to provide UACs for each of the surface areas for use in the analysis. For elevated



Note: Largest areas modeled for each WMU type have been omitted from the chart to improve clarity.



Note: Largest areas modeled for each WMU type have been omitted from the chart to improve clarity.

51

sources, area-height combinations were modeled that best covered the range of area-height combinations found in the database. For any specific WMU, a UAC was estimated using an interpolation routine that used the UACs immediately above and below the actual area of the unit. For elevated sources, the UACs associated with the modeled height closest to the actual WMU height were used. The interpolation routine provides a technique for minimizing the number of ISCST3 runs required while also minimizing the error associated with the difference between the UACs for preselected areas and the UAC for the actual area of the WMU.

Landfills and LAUs were modeled as ground-level area sources while wastepiles and tanks were treated as elevated area sources. Fourteen surface areas were selected for modeling for landfills and LAUs. Twenty-nine surface area-height combinations were selected for wastepiles and 33 area-height combinations for tanks. The areas were selected using a modified version of a statistical method called the Dalenius Hodges procedure (see Appendix A, Volume II for details). This procedure divided into strata the skewed distribution of areas found in the Industrial D survey database so that all WMUs in the database would be adequately represented. The median area in each stratum was then used in the dispersion modeling. This procedure is described in more detail in Section 3 of Volume II.

The selected area-height combinations for the four types of WMUs were modeled with 29 representative meteorological locations in the continental United States to estimate UACs. The 5-year average UACs at all receptor points were calculated for the long-term or chronic exposure scenario. They were used as inputs to the Monte Carlo analysis and as input to the interpolation routine discussed above.

A similar methodology and assumptions were used to model dispersion for acute and subchronic exposures. Since the ISCST3 model uses hourly meteorological data, the outputs from the model can be used to develop any averaging times equal to or greater than 1 hour. One set of ISCST3 runs (for all area-height combinations and 29 meteorological stations) was performed for both acute and subchronic analyses, resulting in 5 years of hourly average concentrations at each receptor. For each area, meteorological location, and receptor location, the maximum air concentration for any 24-hour period over the 5 years was selected for acute exposures. Then, for each area and meteorological station, the maximum 24-hour air concentration among all receptor locations at each distance modeled was selected, and this was used as the UAC for that area and meteorological station for acute exposure. The same method was used to determine the subchronic UAC, except that the maximum 30-day period over the 5 years was used instead of the maximum 24-hour period. It was assumed that the greatest potential for acute exposure would be closest to the site, therefore, the receptors points were placed at 25, 50, and 75 meters from the edge of the WMU, in 16 directions at each distance.

3.2.4 Exposure Modeling/Risk Estimation

The previous sections described how emissions and UACs were developed. This section describes the models used to combine those results with exposure factor distributions to calculate risk or hazard quotient and risk-specific waste concentration (C_w).

For carcinogens, a Monte Carlo analysis was performed in which the location of the receptor and various exposure factors (body weight, inhalation rate, and exposure duration) were

52

varied. For each constituent and WMU type combination, a separate Monte Carlo simulation was run for each WMU in the Industrial D Survey or TSDR Survey database. The emission rate for the specific constituent from the specific WMU was used as an input to the Monte Carlo simulation and was not varied across iterations within a simulation. Approximately 1,000 iterations were performed for each WMU, resulting in a distribution of waste concentrations (C_w) that would result in the specified risk criteria. This distribution captures the range in waste concentration attributable to the variability in potential location and in exposure factors associated with each receptor. From this distribution, the 85th, 90th, and 95th percentiles were selected to characterize the distribution. These percentiles represent the percentage of receptors that are protected at the risk criterion for a specific WMU.

When the Monte Carlo simulation had been run for all the WMUs, a cumulative distribution across all facilities for each protection level (85th, 90th, or 95th percent of receptors) was obtained for each receptor at each distance. This distribution reflects the variability across facilities. In developing this distribution, the results were weighted using the facility weights from the Industrial D Survey data. These weights indicate the number of facilities in the United States represented by a particular facility in the Industrial D Survey database. The resulting cumulative distribution accounts for variability across all facilities represented, not just those actually modeled. The TSDR survey data used to characterize tanks did not include facility weights; therefore, the tank distributions are not weighted.

Hazard quotients for noncarcinogens depend only on air concentration and the health benchmark (a reference concentration). Therefore, exposure factors are not used and of the variables varied in the Monte Carlo analysis, only the location of the receptor is relevant. Because the location of the receptor is such a simple distribution, a Monte Carlo analysis was unnecessary for noncarcinogens; the distribution of hazard quotient (and therefore C_w) based on the distribution of the location of the receptor can be obtained analytically by calculating hazard quotient (and C_w) for each of the 16 receptor locations (or directions around the site) and taking the desired percentiles from those 16 values.

Exposure and risk modeling for subchronic and acute exposures differed somewhat from the modeling for chronic exposures in several respects. All acute and subchronic health benchmarks are analagous to chronic noncarcinogen benchmarks, so exposure factors were not used. In addition, receptor location around the site was not varied for acute and subchronic exposures. Therefore, a Monte Carlo analysis was not performed for subchronic and acute exposures. Instead, a point estimate of C_w was calculated for each WMU in the Industrial D database using the single sector that resulted in the maximum air concentration. This point estimate represents the maximum, or 100th percentile, concentration and therefore is most comparable to the 100th percentile of the distribution generated by the Monte Carlo model for chronic exposures. The point estimate can be interpreted as the level at which 100 percent of receptors are protected at a particular WMU. A distribution across all WMUs of a specific type was generated from these point estimates, and the 90th percentile of that distribution is presented in the results for subchronic and acute exposures.

3.2.4.1 Obtain Health Benchmarks. For chronic exposures, standard health benchmarks (cancer slope factors for carcinogens and reference concentrations for noncarcinogens) were obtained for each constituent (these are shown in Table 3-3). Chronic

53

benchmarks for 15 chemicals were developed explicitly for the Air Characteristic Study. However, a benchmark could not be developed for one chemical (3,4-dimethylphenol) due to lack of appropriate data, so, although EPA has addressed this chemical, risks for it could not be quantified.

Information on acute and intermediate/subchronic inhalation benchmark values and occupational exposure limits was collected for use in the analysis. These data are also shown in Table 3-3.

3.2.4.2 Calculate Risk or Hazard Quotient. The risk or hazard quotient associated with a unit waste concentration was calculated for each iteration based on the calculated air concentration and the exposure factors selected for the iteration.

Carcinogens. Adult receptors modeled include adult residents and off-site workers. Risk for adults is calculated using long-term average air concentration that is constant over the entire exposure duration. The inhalation rate, exposure frequency, and exposure duration differ for residents and workers. Body weight is the same for all adults, whether resident or worker. All exposure factors for adults are held constant over the entire exposure duration.

Calculation of Risk for Carcinogens for Adults

$$Risk_{calc'd} = \frac{C_{air} * CSF * IR * ED * EF}{BW * AT * 365\ d/yr}$$

where

$Risk_{calc'd}$	=	individual risk associated with unit waste concentration (per mg/kg)
C_{air}	=	air concentration associated with a unit waste concentration ([mg/m^3]/[mg/kg])
CSF	=	cancer slope factor (per mg/kg-d)
IR	=	inhalation rate (m^3/d)
ED	=	exposure duration (yr)
EF	=	exposure frequency (d/yr)
BW	=	body weight (kg)
AT	=	averaging time (yr) = 70

Three child age groups, or cohorts, were used to model child exposures: 0 to 3, 4 to 10, and 11 to 18 years of age. These cohorts reflect the age cohorts for which inhalation rate data are available. Results were calculated and saved for receptors falling into each of these three age cohorts at the start of exposure and presented as child 1 (0-3), child 2 (4-10), and child 3 (11-18). An exposure duration was selected randomly for each of the three starting age cohorts from a distribution specific to each starting age cohort. For each age cohort, exposure begins at a starting age selected at random within the cohort and then continues through succeeding age cohorts and into adulthood as necessary until the exposure duration selected for that starting age cohort is reached.

Annual risk for each year of exposure (from starting age to starting age plus exposure duration) was calculated and summed over the exposure duration for each child receptor. If the child reached age 19 before the exposure duration ended, adult exposure factors were used for the remainder of the exposure duration. This approach requires both body weight and inhalation rate distributions by year of age; however, only body weight is available by year. Inhalation rate is available only for the age groups used to define the cohorts (0-3, 4-10, and 11-18 years). Because inhalation rate data could not be disaggregated to individual years of age, we retained

54

Table 3-3. Inhalation Health Benchmarks Used in the Air Characteristic Analysis

CAS #	Name	Chronic RfC (mg/m^3)	Ref[a]	Inhal URF $(\mu g/m^3)^{-1}$	Inhal CSF $(mg/kg/day)^{-1}$	Ref[a]	Subchronic RfC (mg/m^3)	Ref[a]	Acute RfC (mg/m^3)	Ref[a]
75-07-0	Acetaldehyde	9.0E-03	I	2.2E-06	7.7E-03	I	9.0E-02	C		
67-64-1	Acetone	3.1E+01	A	NA	NA		3.1E+01	A	6.2E+01	A
75-05-8	Acetonitrile	6.0E-02	I	NA	NA		6.0E-02	I		
107-02-8	Acrolein	2.0E-05	I	NA	NA		2.0E-04	C	1.1E-04	A
79-06-1	Acrylamide	NA		1.3E-03	4.6E+00	I				
79-10-7	Acrylic acid	1.0E-03	I	NA	NA		3.0E-03	H	6.0E+00	CA
107-13-1	Acrylonitrile	2.0E-03	I	6.8E-05	2.4E-01	I	2.0E-02	C	2.2E-01	A
107-05-1	Allyl chloride	1.0E-03	I	NA	NA		1.0E-02	H		
62-53-3	Aniline	1.0E-03	I	NA	NA		1.0E-02	H		
7440-38-2	Arsenic	NA		4.3E-03	1.5E+01	I			4.0E-04	CA
7440-39-3	Barium	5.0E-04	H	NA	NA		5.0E-03	H		
71-43-2	Benzene	NA		8.3E-06	2.9E-02	I	1.3E-02	A	1.6E-01	A
92-87-5	Benzidine	NA		6.7E-02	2.3E+02	I				
50-32-8	Benzo(a)pyrene	NA		8.8E-04	3.1E+00	N				
7440-41-7	Beryllium	2.0E-05	I	2.4E-03	8.4E+00	I				
75-27-4	Bromodichloromethane	NA		1.8E-05	6.2E-02	D				
75-25-2	Bromoform (Tribromomethane)	NA		1.1E-06	3.9E-03	I				
106-99-0	Butadiene, 1,3-	NA		2.8E-04	1.8E+00	I				
7440-43-9	Cadmium	NA		1.8E-03	6.3E+00	I				
75-15-0	Carbon disulfide	7.0E-01	I	NA	NA		7.0E-01	H	2.0E+01	CA
56-23-5	Carbon tetrachloride	NA		1.5E-05	5.3E-02	I	3.1E-01	A	1.3E+00	A
126-99-8	Chloro-1,3-butadiene, 2- (Chloroprene)	7.0E-03	H	NA	NA		7.0E-02	H		
108-90-7	Chlorobenzene	2.0E-02	H	NA	NA		2.0E-01	SF		

(continued)

55

Table 3-3. (continued)

CAS #	Name	Chronic RfC (mg/m^3)	Ref[a]	Inhal URF $(\mu g/m^3)^{-1}$	Inhal CSF $(mg/kg/day)^{-1}$	Ref[a]	Subchronic RfC (mg/m^3)	Ref[a]	Acute RfC (mg/m^3)	Ref[a]
124-48-1	Chlorodibromomethane	NA		2.4E-05	8.4E-02	D				
67-66-3	Chloroform	NA		2.3E-05	8.1E-02	I	2.4E-01	A	4.9E-01	A
95-57-8	Chlorophenol, 2-	1.4E-03	D	NA	NA					
7440-47-3	Chromium VI	1.0E-04	I	1.2E-02	4.2E+01	I	5.0E-04	A		
7440-48-4	Cobalt	1.0E-05	D	NA	NA		3.0E-05	A		
1319-77-3	Cresols (total)	4.0E-04	D	NA	NA		1.2E-03	C		
98-82-8	Cumene	4.0E-01	I	NA	NA		4.0E+00	C		
108-93-0	Cyclohexanol	2.0E-05	FR	NA	NA		2.0E-04	C		
96-12-8	Dibromo-3-chloropropane, 1,2-	2.0E-04	I	6.9E-07	2.4E-03	H	2.0E-03	C		
95-50-1	Dichlorobenzene, 1,2-	2.0E-01	H	NA	NA		2.0E+00	H		
106-46-7	Dichlorobenzene, 1,4-	8.0E-01	I	NA	NA		2.5E+00	H	4.8E+00	A
75-71-8	Dichlorodifluoromethane	2.0E-01	H	NA	NA		2.0E+00	H		
107-06-2	Dichloroethane, 1,2-	NA		2.6E-05	9.1E-02	I	8.1E-01	C	8.1E-01	A
75-35-4	Dichloroethylene, 1,1-	NA		5.0E-05	1.8E-01	I	7.9E-02	A		
78-87-5	Dichloropropane, 1,2-	4.0E-03	I	NA	NA		1.3E-02	H	2.3E-01	A
10061-01-5	Dichloropropene, cis-1,3-	2.0E-02	I	3.7E-05	1.3E-01	H	2.0E-02	H		
10061-02-6	Dichloropropene, trans-1,3-	2.0E-02	I	3.7E-05	1.3E-01	H	2.0E-02	H		
57-97-6	Dimethylbenz(a)anthracene, 7,12-	NA		2.4E-02	8.4E+01	D				
68-12-2	Dimethylformamide, *N,N-*	3.0E-02	I	NA	NA		3.0E-02	H		
95-65-8	Dimethylphenol, 3,4-	NA		NA	NA					
121-14-2	Dinitrotoluene, 2,4-	NA		1.9E-04	6.8E-01	D				
123-91-1	Dioxane, 1,4-	8.0E-01	D	NA	NA		8.0E+00	C	6.0E+00	CA
122-66-7	Diphenylhydrazine, 1,2-	NA		2.2E-04	7.7E-01	I				
106-89-8	Epichlorohydrin	1.0E-03	I	1.2E-06	4.2E-03	I	1.0E-02	H	3.0E+00	CA
106-88-7	Epoxybutane, 1,2-	2.0E-02	I	NA	NA		2.0E-01	C		

(continued)

56

Table 3-3. (continued)

CAS #	Name	Chronic RfC (mg/m^3)	Ref[a]	Inhal URF $(\mu g/m^3)^{-1}$	Inhal CSF $(mg/kg/day)^{-1}$	Ref[a]	Subchronic RfC (mg/m^3)	Ref[a]	Acute RfC (mg/m^3)	Ref[a]
111-15-9	Ethoxyethanol acetate, 2-	3.0E-01	D	NA	NA				3.0E-01	CA
110-80-5	Ethoxyethanol, 2-	2.0E-01	I	NA	NA		2.0E+00	H	9.0E-01	CA
100-41-4	Ethylbenzene	1.0E+00	I	NA	NA		8.7E-01	A		
106-93-4	Ethylene dibromide	2.0E-04	H	2.2E-04	7.7E-01	I	2.0E-03	H		
107-21-1	Ethylene glycol	6.0E-01	D	NA	NA		6.0E+00	C	1.3E+00	A
75-21-8	Ethylene oxide	NA		1.0E-04	3.5E-01	H	1.6E-01	A		
50-00-0	Formaldehyde	NA		1.3E-05	4.6E-02	I	1.2E-02	A	6.1E-02	A
98-01-1	Furfural	5.0E-02	H	NA	NA		5.0E-01	H		
87-68-3	Hexachloro-1,3-butadiene	NA		2.2E-05	7.7E-02	I				
118-74-1	Hexachlorobenzene	NA		4.6E-04	1.6E+00	I				
77-47-4	Hexachlorocyclopentadiene	7.0E-05	H	NA	NA		7.0E-04	H		
67-72-1	Hexachloroethane	NA		4.0E-06	1.4E-02	I	5.8E+01	A	5.8E+01	A
110-54-3	Hexane, –	2.0E-01	I	NA	NA		2.0E-01	H		
78-59-1	Isophorone	1.2E-02	FR	NA	NA		1.2E-01	C		
7439-96-5	Manganese	5.0E-05	I	NA	NA		5.0E-04	C		
7439-97-6	Mercury	3.0E-04	I	NA	NA		3.0E-04	H	2.0E-03	CA
67-56-1	Methanol	1.3E+01	D	NA	NA				3.0E+01	CA
110-49-6	Methoxyethanol acetate, 2-	3.0E-02	D	NA	NA					
109-86-4	Methoxyethanol, 2-	2.0E-02	I	NA	NA		2.0E-01	H	2.0E-02	CA
74-83-9	Methyl bromide (Bromomethane)	5.0E-03	I	NA	NA		1.9E-01	A	1.9E-01	A
74-87-3	Methyl chloride (Chloromethane)	NA		1.8E-06	6.3E-03	H	4.1E-01	A	1.0E+00	A
78-93-3	Methyl ethyl ketone	1.0E+00	I	NA	NA		1.0E+00	H	3.0E+01	CA
108-10-1	Methyl isobutyl ketone	8.0E-02	H	NA	NA		8.0E-01	H		
80-62-6	Methyl methacrylate	7.0E-01	I	NA	NA		7.0E+00	C		
1634-04-4	Methyl tert-butyl ether	3.0E+00	I	NA	NA		2.1E+00	A	6.1E+00	A

(continued)

57

Table 3-3. (continued)

CAS #	Name	Chronic RfC (mg/m^3)	Ref[a]	Inhal URF $(\mu g/m^3)^{-1}$	Inhal CSF $(mg/kg/day)^{-1}$	Ref[a]	Subchronic RfC (mg/m^3)	Ref[a]	Acute RfC (mg/m^3)	Ref[a]
56-49-5	Methylcholanthrene, 3-	NA		2.1E-03	7.4E+00	D				
75-09-2	Methylene chloride	3.0E+00	H	4.7E-07	1.6E-03	I	3.0E+00	H	1.0E+01	A
91-20-3	Naphthalene	3.0E-03	I	NA	NA		3.0E-02	C		
7440-02-0	Nickel	NA		2.4E-04	8.4E-01	I	2.0E-03	C	1.0E-02	CA
98-95-3	Nitrobenzene	2.0E-03	H	NA	NA		2.0E-02	H		
79-46-9	Nitropropane, 2-	2.0E-02	I	2.7E-03	9.4E+00	H	2.0E-02	H		
55-18-5	Nitrosodiethylamine	NA		4.3E-02	1.5E+02	I				
924-16-3	Nitrosodi-*n*-butylamine	NA		1.6E-03	5.6E+00	I				
930-55-2	*N*-Nitrosopyrrolidine	NA		6.1E-04	2.1E+00	I				
108-95-2	Phenol	6.0E-03	FR	NA	NA		6.0E-02	C	6.0E+00	CA
85-44-9	Phthalic anhydride	1.2E-01	H	NA	NA		1.2E-01	H		
75-56-9	Propylene oxide	3.0E-02	I	3.7E-06	1.3E-02	I	3.0E-02	H	6.0E+00	CA
110-86-1	Pyridine	7.0E-03	O	NA	NA					
100-42-5	Styrene	1.0E+00	I	NA	NA		3.0E+00	H	2.0E+01	CA
1746-01-6	TCDD, 2,3,7,8-	NA		3.3E+01	1.6E+05	H				
630-20-6	Tetrachloroethane, 1,1,1,2-	NA		7.4E-06	2.6E-02	I				
79-34-5	Tetrachloroethane, 1,1,2,2-	NA		5.8E-05	2.0E-01	I	2.7E+00	A		
127-18-4	Tetrachloroethylene	3.0E-01	A	5.8E-07	2.0E-03	SF	3.0E+00	C	1.4E+00	A
108-88-3	Toluene	4.0E-01	I	NA	NA		4.0E+00	C	1.5E+01	A
95-53-4	Toluidine, *o*-	NA		6.9E-05	2.4E-01	D				
76-13-1	Trichloro-1,2,2-trifluoroethane, 1,1,2-	3.0E+01	H	NA	NA		3.0E+01	H		
120-82-1	Trichlorobenzene, 1,2,4-	2.0E-01	H	NA	NA		2.0E+00	H		
71-55-6	Trichloroethane, 1,1,1-	1.0E+00	SF	NA	NA		3.8E+00	A	1.1E+01	A
79-00-5	Trichloroethane, 1,1,2-	NA		1.6E-05	5.6E-02	I				
79-01-6	Trichloroethylene	NA		1.7E-06	6.0E-03	SF	5.4E-01	A	1.1E+01	A

(continued)

58

Table 3-3. (continued)

CAS #	Name	Chronic RfC (mg/m^3)	Ref[a]	Inhal URF $(\mu g/m^3)^{-1}$	Inhal CSF $(mg/kg/day)^{-1}$	Ref[a]	Subchronic RfC (mg/m^3)	Ref[a]	Acute RfC (mg/m^3)	Ref[a]
75-69-4	Trichlorofluoromethane	7.0E-01	H	NA	NA		7.0E+00	H		
121-44-8	Triethylamine	7.0E-03	I	NA	NA		7.0E-02	C		
7440-62-2	Vanadium	7.0E-05	D	NA	NA		7.0E-05	C	7.0E-04	C
108-05-4	Vinyl acetate	2.0E-1	I	NA	NA		2.0E-01	H		
75-01-4	Vinyl chloride	NA		8.4E-05	3.0E-01	H	7.7E-02	A	1.3E+00	A
1330-20-7	Xylenes (total)	4.0E-01	A	NA	NA		3.0E+00	A	4.3E+00	A

CAS = Chemical Abstract Service.
CSF = Cancer slope factor.
NA = Not available.
RfC = Reference concentration.
URF = Unit risk factor.

[a]References:
I = IRIS (U.S. EPA, 1999)
H = HEAST (U.S. EPA, 1997a)
A = Agency for Toxic Substances Disease Registry; minimal risk levels (MRLs) (ATSDR, 1999)
SF = Superfund Risk Issue Paper (U.S. EPA, 1996, U.S. EPA, nd)
FR = 63 FR 64371-402 (U.S. EPA, 1998)
N = NCEA Risk Assessment Issue Paper (U.S. EPA, 1994b)
D = Developed for this study
O = Other source (see Volume II, Sections 6.1.1 and 6.1.2)
C = Calculated from chronic RfC value
CA = Cal EPA 1-h acute inhalation reference exposure levels (RELs) (CalEPA, 1998)
E = Acute exposure guideline level (AEGL) (U.S. EPA, 1997b)

59

year-by-year body weights and used the inhalation rate for the cohort associated with each year of age for that year. Thus, the inhalation rate is a constant for all ages within an age cohort and changes only when the receptor ages from one cohort to the next. Both EPA and a statistician experienced in working with EFH exposure factor data (L. Myers, RTI, personal communication with Anne Lutes, RTI, March 16, 1998) preferred this approach over the alternative of pooling body weights to the age cohort age ranges because it retains the most detail from the available data without sacrificing statistical rigor.

Noncarcinogens. Because the hazard quotient equation for noncarcinogens does not consider exposure factors, there is no difference in results for different receptors at the same location (e.g., adult resident, child resident, and offsite worker). Therefore, only an adult resident was modeled for noncarcinogens.

Calculation of Hazard Quotient for Noncarcinogens

$$HQ_{calc'd} = \frac{C_{air}}{RfC}$$

where

$HQ_{calc'd}$ = hazard quotient associated with unit waste concentration (per mg/kg)

C_{air} = air concentration associated with a unit waste concentration ([mg/m^3]/[mg/kg])

RfC = reference concentration (mg/m^3)

When a particular constituent had both carcinogenic and noncarcinogenic effects, the carcinogenic risk was used, because it is generally more protective.

3.2.4.3 Risk-Specific Waste Concentration. The final step in each iteration was to backcalculate the risk-specific waste concentration from the risk or hazard quotient corresponding to a unit waste concentration. Because risk is linear with respect to waste concentration in the models used in this analysis, this may be done by a simple ratio technique.

As mentioned, risk is assumed to be linear with waste concentration. The assumption of linearity is accurate for the dispersion modeling and the exposure and risk modeling. However, the emissions model is linear only for land-based units and for tanks with no biodegradation; for tanks with biodegradation, the emissions model is nonlinear with respect to biodegradation. At low concentrations, biodegradation in tanks is first order. However, at concentrations in excess of the half-saturation level, biodegradation becomes zero order. In order to address this, emissions were modeled in the aqueous phase at 0.001 mg/L to capture first-order biodegradation and at the solubility to capture zero-order biodegradation. These emission rates then were normalized to a unit concentration by dividing by 0.001 or the solubility. When the backcalculated waste concentration based on first-order biodegradation exceeded the half-saturation constant, suggesting that biodegradation would be zero order, it was recalculated based on the normalized solubility limit emission rate.

The results for all WMU types presented in Section 4 and Volume III were calculated as described above using aqueous-phase emission rates. Most of the waste streams managed in the types of units modeled are expected to contain constituents in the aqueous, rather than the organic, phase; therefore, this is the most realistic scenario. However, results based on organic-phase emissions are of interest in two circumstances: when organic-phase emissions are higher than aqueous-phase emissions, and when backcalculated results based on aqueous-phase

emissions exceed physical limitations on the aqueous phase, such as the soil saturation concentration or solubility. These are discussed below.

Most chemicals are better able to volatilize from an aqueous medium than from an organic medium; therefore, for most chemicals, the aqueous-phase emission rates are considerably higher than organic-phase emission rates. However, for a few chemicals (most notably formaldehyde), the organic-phase emissions are higher than the aqueous-phase emissions and protective waste concentrations based on organic-phase emissions would be lower than protective waste concentrations based on aqueous-phase emissions. When this is the case, the results based on aqueous-phase emissions are footnoted to indicate this. This does not invalidate the aqueous-phase results, as that is still the most likely waste matrix.

Some of the backcalculated waste concentrations based on aqueous-phase emissions exceed the soil saturation concentration or solubility under standard conditions. These are the theoretical maximum possible aqueous-phase concentration in soil or water, respectively; once this is exceeded, free (organic-phase) product will occur in the soil or wastewater. In tanks, free organic phase product will either sink, yielding aqueous-phase emissions from a concentration equal to the solubility, or float on the surface, yielding emissions from the organic phase at a concentration of pure component. The soil saturation concentration and solubility under standard temperature and pH conditions (20-25 C and neutral pH) have been estimated for each chemical in the analysis, but these are somewhat site- and waste-specific values. Therefore, a backcalculated concentration may exceed them in some situations but not in others. See Section 7.10.2 of Volume II for details on how the soil saturation concentration and solubility were estimated. When the backcalculated concentration based on aqueous-phase emissions exceeded the typical soil saturation concentration or solubility calculated for this analysis, the result was footnoted to indicate whether pure component (i.e., a concentration of 10^6 mg/kg or

Modifications to Methodology for Lead. Human health risk assessment for lead is unique. Instead of developing an RfC in the traditional manner, all identified sources of lead exposure (including background) are used to predict blood lead (PbB) levels in the exposed individuals. The predicted PbB levels are compared to a target PbB. PbB levels have long been used as an index of body lead burdens and as an indicator of potential health effects.

The Integrated Exposure Uptake Biokinetic Model (IEUBK) (U.S. EPA, 1994a) was developed to predict PbB levels for an individual child or a population of children. The model was specifically designed to evaluate lead exposure in young children (birth to 7 years of age) because this age group is known to be highly sensitive to lead exposure. Therefore, only two receptors were modeled for lead: children aged 0 to 3 years and 3 to 7 years. Adults (including workers) and older children were excluded from the analysis for lead because those age groups are considered less sensitive to lead than 0- to 7-year-olds (and, in fact, the pharmacokinetic relationships in the IEUBK model are only valid for 0- to 7-year-olds).

For this analysis, the IEUBK model was used to identify air concentrations that would result in a less than 5 percent probability of having a PbB level higher than the target PbB. That concentration in air was then used in place of an RfC in the calculations. Because the IEUBK model cannot be run in a backcalculation mode, different air concentrations were modeled until one was found that satisfied the 95 percent protection level desired. A target blood lead level of 10 µg/dL was selected because that level has been identified as a level of concern by the Centers for Disease Control (CDC) (U.S. EPA, 1994a).

The IEUBK model inputs are summarized in Volume II. They are inhalation rate, body weight, media concentrations (including soil, indoor dust, water, and food), and indoor air concentration as a percentage of outdoor air concentration.

mg/L) would result in a risk exceeding the target risk when modeled using organic-phase emission rates.

3.3 Analysis of Variability and Uncertainty

The purpose of this section is to discuss the methods that are used in this study to capture variability and uncertainty. Variability and uncertainty are discussed separately because they are fundamentally different.

This discussion describes the treatment of variability in some parameters used to describe human receptors and their behavior. Treatment of variability using a Monte Carlo simulation forms the basis for the risk distributions. Uncertainty necessitates the use of assumptions, default values, and imputation techniques in this study. Table 3-4 presents the major categories of variability and uncertainty and how they have been addressed in this study. The columns in the table show scenario uncertainty, model uncertainty, parameter uncertainty, and parameter variability. The rows present the five main model components in the analysis: source characterization, the emissions model, the dispersion model, the exposure model, and the risk model.

Variability arises from true heterogeneity in characteristics such as body weight differences within a population or differences in contaminant levels in the environment.

Uncertainty represents lack of knowledge about factors, such as the nature of adverse effects from exposure to constituents, which may be reduced with additional research.

3.3.1 Variability

In conducting a national risk assessment, numerous parameters will vary across the nation. Variability is often used interchangeably with the term "uncertainty," but this is not strictly correct. Variability is tied to variations in physical, chemical, and biological processes and cannot be reduced with additional research or information. Although variability may be known with great certainty (e.g., age distribution of a population may be known and represented by the mean age and its standard deviation), it cannot be eliminated and needs to be treated explicitly in the analysis. Spatial and temporal variability in parameter values used to model exposure and risk account for the distribution of risk in the exposed population.

In planning this analysis, it was important to specifically address as much of the variability as possible, either directly in the Monte Carlo analysis or through disaggregation of discrete parts of the analysis. For example, use of a refined receptor grid accounts for spatial variability in concentrations around an WMU. Variability in WMU characteristics is accounted for using a large database of individual WMUs that represent the range of possible WMU characteristics.

3.3.2 Uncertainty

Uncertainty is a description of the imperfection in knowledge of the true value of a particular parameter. In contrast to variability, uncertainty is reducible by additional information

42

Table 3-4. Summary of Variability and Uncertainty in the Study

	Scenario Uncertainty	Model Uncertainty	Parameter Uncertainty	Parameter Variability
Source Characterization	Ancillary site operations not addressed (e.g., emissions from truck traffic or unloading operations)		Quality of survey data not addressed (e.g., age and representativeness of data, missing data) Imputation of parameter values not directly surveyed based on statistical inference using data for similar WMUs	Facility-specific location, waste volume, dimensions, engineering design parameters used to address variability in WMU parameters Biodegradtion variations with site-specific factors not addressed
Emissions Model	Variations in operation practices not addressed	Instantaneous release model used for acute and subchronic peak releases Competing release mechanisms (e.g., runoff, erosion, leaching) not addressed	Dependencies of biodegradation, volatility, and temperature addressed through sensitivity analysis and use of seasonal temperature variations Differences in biodegradation rates between soil and aqueous systems not addressed	Facility-specific locations and meteorology used to address variability in WMU parameters
Dispersion Model		Model error increased by about 2% by not using wet deposition/depleti on option Photochemical reactions and degradation not addressed 30-day and 1-day averages used for subchronic and acute exposures, respectively	Sensitivity analyses conducted on a number of parameters including shape and orientation of WMU, meteorologic data, and receptor grid	29 meteorologic stations used to represent climate regions 14 surface areas used to represent distribution of surface area for landfills, LAUs 29 surface areas/ heights combinations used for wastepiles 33 surface areas/height combinations used for tanks
Exposure Model	Indirect exposures not addressed		Sensitivity analysis conducted for receptor grid	16 receptor locations at each distance used in Monte Carlo analysis Exposure factor distributions developed and used in Monte Carlo analysis
Risk Model			Health benchmark uncertainty not addressed (e.g., high to low dose extrpolation, animal to human extrapolation)	Variabilility in individual dose response not addressed

43

gathering or analysis activities (better data, better models). EPA typically classifies the major areas of uncertainty in risk assessments as scenario uncertainty, model uncertainty, and parameter uncertainty. Scenario uncertainty refers to missing or incomplete information needed to fully define exposure and dose. Model uncertainty is a measure of how well the model simulates reality. Finally, parameter uncertainty is the lack of knowledge regarding the true value of a parameter that is used in the analysis. While some aspects of uncertainty were directly addressed in the analysis, much of the uncertainty associated with this analysis could only be addressed qualitatively.

Sources of scenario uncertainty include the assumptions and modeling decisions that are made to represent an exposure scenario. Because we lack information or resources to define and model actual exposure conditions, uncertainty is introduced into the analysis. Despite the complexity of this analysis, it was necessary to exclude or simplify actual exposure conditions. For example, this analysis only addresses inhalation exposures; indirect exposure pathways were excluded. Professional judgement, often coupled with using the results of sensitivity analysis, is used to decide which parameters to include in describing exposure conditions and behaviors. These judgements are imperfect and uncertainty is introduced.

To reduce model uncertainty, EPA generally selected models that are considered state-of-the-art. Model uncertainty is associated with all models used in all phases of a risk assessment. These include the animal models used as surrogates for testing human carcinogenicity, dose-response models used in extrapolations, and computer models used to predict the fate and transport of chemicals in the environment. Computer models are simplifications of reality, requiring exclusion of some variables that influence predictions but cannot be included in models due either to increased complexity or to a lack of data on a particular parameter. The risk assessor needs to consider the importance of excluded variables on a case-by-case basis because a given variable may be important in some instances and not in others. A similar problem can occur when a model that is applicable under average conditions is used when conditions differ from the average. In addition, choosing the correct model form is often difficult when conflicting theories seem to explain a phenomenon equally well. Modeling uncertainty is not addressed directly in this study but is discussed qualitatively.

Parameter uncertainty occurs when (1) there is a lack of data about the parameters used in the equations, (2) the data that are available are not representative of the particular instance being modeled, or (3) parameter values cannot be measured precisely and/or accurately either because of equipment limitations or because the quantity being measured varies spatially or temporally. Random, or sample errors, are a common source of parameter uncertainty that is especially critical for small sample sizes. More difficult to recognize are nonrandom or systematic errors that result from bias in sampling, experimental design, or choice of assumptions.

4.0 Summary of Risk Assessment Results

This section presents an overview of the results of the risk analysis that evaluated the direct inhalation risks from waste management unit (WMU) emissions. These results present waste concentration levels (C_w's) that protect 90 percent of receptors at distances of 25, 150, and 1,000 m from the edge of the WMU across 90 percent of the sites (90/90 protection levels) at a risk level of 10^{-5} or an HQ of 1. This subset of the results was selected for presentation purposes only and does not imply that these are the results that would be used for an air characteristic. The detailed results that are summarized here, as well as results for alternative risk levels, additional distances, and additional protection levels, are presented in Volume III, *Results*, on CD-ROM.

4.1 Overview of Results

The most protective (i.e., lowest) 90/90 C_w values for adults across all WMUs ranged from 0.005 ppm to 1 million ppm across all chemicals modeled. The lowest value, 0.005 ppm, was for 2,3,7,8-TCDD in tanks. It should be noted that this value exceeds TCDD's solubility limit (0.00002 ppm) at a neutral pH and temperature of 25°C. As discussed in Section 3.2.4.3, the solubility limit (or soil saturation concentration for land-based units) is a site-specific value, as it varies with pH and temperature. Due to this uncertainty, all the results in this report are shown with the solubility or soil saturation concentration at neutral pH and a temperature of 20-25°C only for comparison purposes. The value for the next lowest chemical (Nitrosodiethylamine) was 0.1 ppm. Chemicals with a C_w of 1 million ppm did not have any concentration that would meet the specified risk level of 10^{-5} or HQ =1.

Figures 4-1 through 4-6 show the number of chemicals at a receptor distance of 150 m with C_w in each order of magnitude range from 0.001 to 1 million ppm for aerated treatment tanks, nonaerated treatment tanks, storage tanks, landfills, LAUs, and wastepiles, respectively. For tanks and landfills, only chronic exposures were modeled, and these results are shown. For LAUs and wastepiles, subchronic and acute results are also shown. Of the 104 chemicals modeled,[1] over half of those less then 1 million ppm fall in the 10 to 10,000 ppm range for tanks and in the 100 to 100,000 ppm range for land-based units. From Figures 4-1 and 4-2 it appears that at least 7 of the 105 chemicals in this study may present a significant potential risk via inhalation at very low concentrations (i.e., <1 ppm) when managed in treatment tanks, and another 28 may be of concern at relatively low concentrations (i.e., < 100 ppm). Figures 4-3 to 4-6 suggest that few chemicals are of concern at low levels (e.g. < 100 ppm) when managed in other WMUs.

[1] Note that 1 chemical of the original 105—3,4-dimethylphenol—was addressed, but risks could not be quantified because data were inadequate to develop a health benchmark for it.

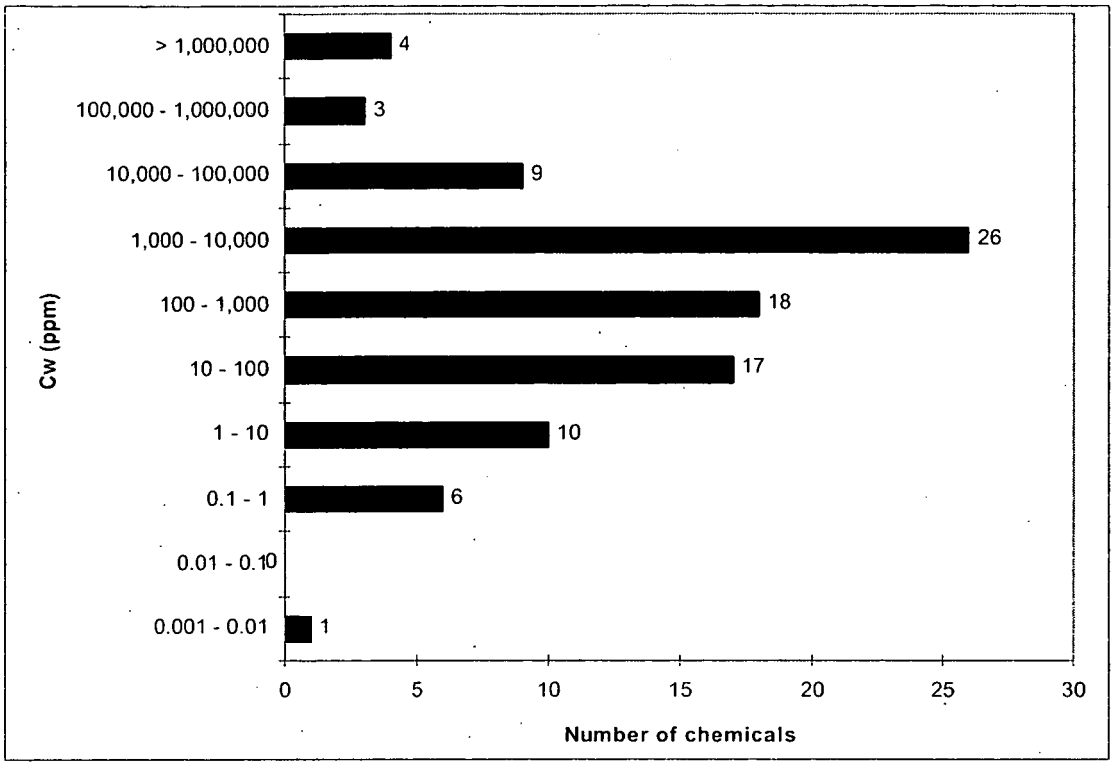


Figure 4-1. Histogram of most protective 90/90 C_W for aerated treatment tanks.

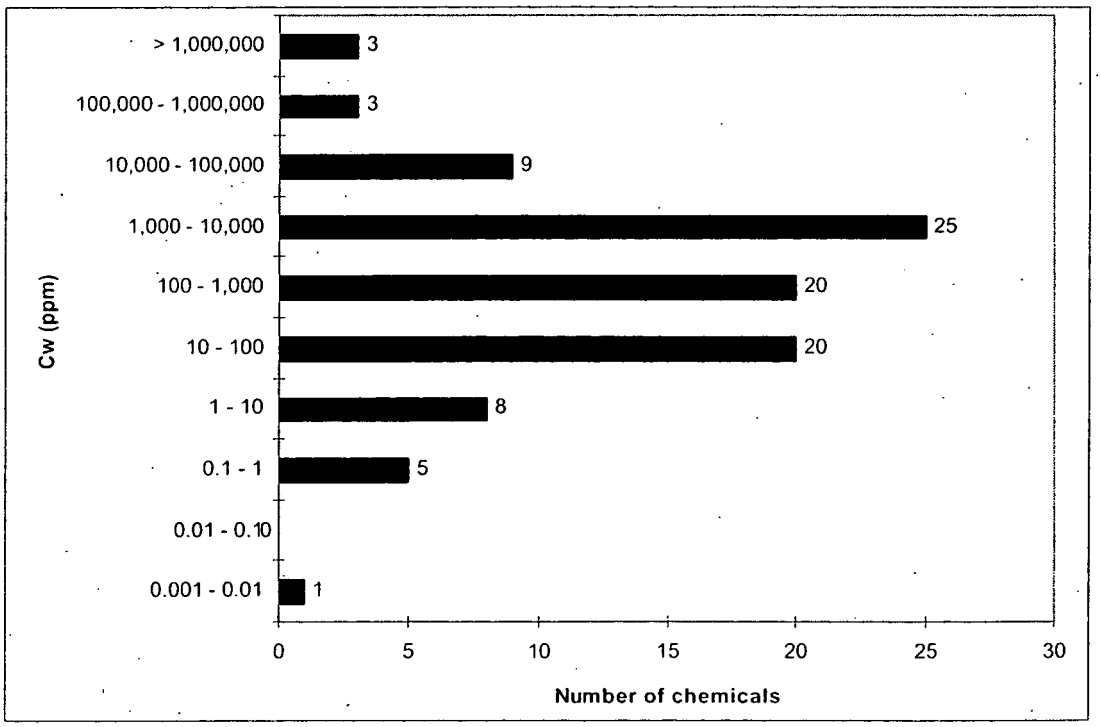


Figure 4-2. Histogram of most protective 90/90 C_W for nonaerated treatment tanks.

4.0 Source Emission Estimates

This chapter describes the source-specific emission model and assumptions used to develop the emission estimates for each waste management unit type. Section 4.1 discusses the selection of a general volatilization model and a particulate emission model to use for the emission estimates. Section 4.2 describes some of the critical model input parameters required to run the volatilization and particulate emission models. Section 4.3 describes modifications made to CHEMDAT8 for land-based units. Subsequent sections describe unit-specific modeling scenarios and assumptions used for the volatilization model effort. The final section of this chapter describes the particulate emission model estimates. Both volatile and particulate emissions were estimated for landfills, land application units, and wastepiles (referred to as land-based WMUs), while only volatile emissions were estimated for tanks.

4.1 Model Selection

4.1.1 Volatile Emission Model Selection

Several factors were considered in selecting emission models for assessing the potential for contaminant exposure through inhalation. In developing acceptable contaminant limits for wastes, the ideal emission model would provide emission estimates that are as accurate as possible without underestimating the contaminant emissions. Because both volatile emissions (for all WMU types) and particulate emissions due to wind erosion (for land-based WMUs) were required in the risk analysis, the volatile emission model had to estimate both volatile emission rates and long-term average soil concentration in the unit (for land-based WMUs). Ideally, the model would provide a relatively consistent modeling approach (in terms of model complexity and accuracy) for each of the different emission sources under consideration. Additionally, the emission model would have to be reviewed both internally by EPA and externally by both state and local agencies and industry representatives. Finally, the model would have to be publicly available for use in more site-specific evaluations.

Based on these considerations, EPA's CHEMDAT8 model was selected as the model to estimate volatile emission rates and long-term average soil concentrations in the WMU. The CHEMDAT8 model was originally developed in projects funded by EPA's Office of Research and Development (ORD) and Office of Air Quality Planning and Standards (OAQPS) to support National Emission Standards for Hazardous Air Pollutants (NESHAPs) from sources such as tanks, surface impoundments, landfills, wastepiles, and land application units for a variety of industry categories including chemical manufacturers, pulp and paper manufacturing, and petroleum refining. It also has been used to support the emissions standards for hazardous waste treatment, storage, and disposal facilities (U.S. EPA, 1991) regulated under Subpart CC rules of RCRA, as amended in 1984. The CHEMDAT8 model is publicly available and has undergone

67

extensive review by both EPA and stakeholder representatives. The CHEMDAT8 spreadsheet model and model documentation may be downloaded at no charge from EPA's web page (http://www.epa.gov/ttn/chief/software.html).

The CHEMDAT8 model considers most of the competing removal pathways that might limit air emissions, including adsorption, hydrolysis (for tanks only), and biodegradation. Adsorption/absorption is the tendency of a chemical or liquid media to attach or bind to the surface or fill the pores of particles in the soil or waste and therefore not volatilize into the air. This tendency to adsorb to or absorb in particles is an important process for estimating the concentration of the chemical on particles emitted to the air due to wind erosion. CHEMDAT8 in its original form models adsorption for land-based units by presuming that the entered waste concentration is in liquid phase. Because waste concentrations are more typically measured as total concentration (liquid plus solid phase), CHEMDAT8 was modified to model adsorption explicitly for an entered total waste concentration for land-based units. Biodegradation is the tendency of a chemical to be broken down or decomposed into less complex chemicals by organisms in the waste or soil. Similarly, hydrolysis is the tendency of a chemical to be broken down or decomposed into less complex chemicals by reaction with water. Chemicals that decompose due to either biodegradation or hydrolysis have lower potential for emission to the air as gases or particles than those that do not. Loss of contaminant by leaching or runoff is not included in the CHEMDAT8 model. Both leaching and runoff are a function of a chemical's tendency to become soluble in water and follow the flow of water (e.g., due to rainfall) down through the soil to groundwater (leaching) or downhill to surface water (runoff). These two mechanisms would also result in less chemical being available for emission to the air as gases or particles. As such, CHEMDAT8 is considered to provide reasonable to slightly high (environmentally protective) estimates of air emissions from the land-based units.

The CHEMDAT8 model was used to estimate the emissions for all WMUs with some modifications. CHEMDAT8 calculates a fraction of chemical emitted. Some additional equations, which are described in Sections 4.4 through 4.7, were added to calculate emission rates in g/m^2–sec and remaining concentration in mg/kg from the fraction emitted and other inputs. This document does not present the equations used by CHEMDAT8 to calculate fraction emitted other than to show modifications made to model adsorbtion for total waste concentration instead of just liquid-phase waste concentration. The reader interested in the CHEMDAT8 algorithms is referred to the CHEMDAT8 documentation (U.S. EPA, 1994e). Additionally, certain equations were modified to prevent division by zero when certain volatilization parameters (Henry's law constant or vapor pressure) were zero (e.g., for metals).[1]

[1] Specifically, the CHEMDAT8 model was modified to prevent division by zero as follows:

- "If-statements" were added to set the biodegradation rate to a negligible level or zero and prevented division by zero when no biodegradation rate constants were available.
- "If-statements" were added to prevent division by zero for chemicals that did not have vapor pressure, Henry's law constant, or diffusivity inputs (e.g., metals).

68

4.1.2 Particulate Emission Model Selection

The model selection criteria for the particulate emission models were similar to those for the volatilization model. Specifically, the particulate emission model would provide as accurate emission estimates as possible without underestimating the contaminant emissions. The model would provide a relatively consistent modeling approach (in terms of model complexity and accuracy) for each of the different emission sources under consideration, and the emission model would have to be both reviewed and publicly available for use for more site-specific evaluations. Two different models were selected to model wind erosion: one for wastepiles (elevated sources) and one for landfills and land application units (ground-level sources). Based on the considerations above, the Cowherd model (U.S. EPA 1985b, 1988) was selected for modeling wind erosion emissions from ground-level sources, and the AP-42 model for wind erosion from aggregate storage piles (U.S. EPA, 1985a) was selected for modeling wind erosion emissions from wastepiles. Newer versions of both of these models are available; however, the newer versions are event-based algorithms that require extensive site-specific data that were not available for the sites modeled in this analysis. The versions used probably result in somewhat higher particulate emission estimates than the event-based algorithms would. This overestimation of particulate emissions is not significant for volatile chemicals, as particulate emissions were found to be a negligible fraction (less than 2 percent in most cases) of total emissions for the volatile chemicals modeled in land-based units. The protective waste concentrations (C_w's) for metals other than mercury (which do not volatilize and are therefore based solely on particulate emissions) may be somewhat lower as a result of this overestimation of emissions.

4.2 Emission Model Input Parameters

This section discusses the various parameters that impact the estimated volatilization and particulate emission rates. Inputs that influence these rates include input parameters specific to the physical and chemical properties of the constituent being modeled, the physical and chemical characteristics of the waste material being managed, input parameters specific to the process and operating conditions of the WMU being modeled, and meteorological parameters.

A general discussion of the physical and chemical properties of the constituents is provided in Section 4.2.1. Critical input parameters for the remaining sets of inputs are discussed first for land-based WMUs and then for tanks. A sensitivity analysis was performed for the 1998 Air Characteristic Study to better understand the impact of certain modeling assumptions on the model results. While the models and data have changed somewhat, those changes would not alter the conclusions drawn; therefore, these sensitivity analyses are not included here. The interested reader is referred to Appendix C of the May 1998 Air Characteristic Study.

4.2.1 Chemical-Specific Input Parameters

Key chemical-specific input parameters include: air-liquid equilibrium partitioning coefficient (vapor pressure or Henry's law constant), liquid-solid equilibrium partitioning coefficient (log octanol-water partition coefficient for organics), biodegradation rate constants, and liquid and air diffusivities. The Hazardous Waste Identification Rule (HWIR) chemical properties database (RTI, 1995; U.S. EPA, 1995b) was used as the primary data source for the

physical and chemical properties for the constituents being modeled. This chemical properties database provided the following chemical-specific input parameters: molecular weight, vapor pressure, Henry's law constant, solubility, liquid and air diffusivities, and log octanol-water partition coefficient. Soil biodegradation rate constants were obtained from Howard et al. (1991). The CHEMDAT8 chemical properties database (U.S. EPA, 1994b) was used as a secondary data source for the physical and chemical properties for the constituents being modeled. This chemical properties database provided the following chemical-specific input parameters: density, boiling point, Antoine's coefficients (to adjust vapor pressure for different temperatures), and biodegradation rate constants for tanks. The biodegradation rate constants in the downloaded CHEMDAT8 database file were compared with the values reported in the summary report that provided the basis for the CHEMDAT8 tank biodegradation rate values (Coburn et al., 1988). Tank biodegradation rate constants for compounds with no data were assigned biodegradation rates equal to the most similar compound in the biodegradation rate database (or set to zero for metals). The specific chemical properties input database used for emission modeling is provided in Appendix B.

4.2.2 Critical Input Parameters for Land-Based WMU Emission Models

4.2.2.1 Volatile Emissions and Waste Concentration. The input parameters used for the CHEMDAT8 land-based unit emissions model are presented in Table 4-1. (Note: The data entry form in the CHEMDAT8 model refers to oil rather than waste; the term waste is used here for clarity.) Of these parameters, two are actually flags to determine which model equations to apply (Input ID Nos. L7 and L9). The most important flag for emission estimates is probably the aqueous waste flag (Input ID No. L7). This flag tells the CHEMDAT8 model which equilibrium partitioning model to use between the liquid and gas phases. For organic wastes, the model uses Raoult's law and the liquid-to-air partition coefficient becomes proportional to the contaminant's partial vapor pressure. For aqueous wastes, the model uses Henry's law and the liquid-to-air partition coefficient becomes proportional to the contaminant's Henry's law coefficient. All land-based WMUs were run twice; once assuming unit concentration (concentration set to 1 mg/kg, assuming Henry's law applies) and once assuming pure component (concentration set to 1E+6 mg/kg, assuming Raoult's law applies). The results presented in Volumes I and III are based on the aqueous phase emission rates (unit concentration and Henry's law). The pure component emission rates were used only to identify chemicals for which greater emissions occur from the organic phase than from the aqueous phase (which is rare) or to identify chemicals for which the aqueous-based results exceeded soil saturation concentrations, and note for these whether the target risk or hazard quotient would be exceeded modeling pure component.

The annual waste quantity is a critical source (site-specific input) parameter. This parameter along with assumptions concerning the frequency of contaminant addition and the dimensions of the unit combine to influence a number of model input parameters (Input ID Nos. L1, L2, L3, L8, and L12).

The CHEMDAT8 model is insensitive to windspeeds for long-term emission estimates from land-based units. Temperature affects the air diffusivity, which affects the volatilization rate and potentially affects the biodegradation rate (biodegradation rates were independent of temperature above 5°C and were set to zero below 5°C). Consequently, temperature is the only

70

Table 4-1. CHEMDAT8 Land-Based Unit Model Input Requirements

Input ID No.	Input Parameter	Data Source/Assumption
L1	Loading (g waste/cm^3 soil)	Waste quantity and/or density from Ind D Survey
L2	Concentration in waste (ppmw)	1 for unit concentration run; 1E+6 for pure component run
L3	Depth of tilling (or unit) (cm)	Assumed or set by capacity
L4	Total porosity	Assumed default value of 0.5
L5	Air porosity (0 if unknown)	Assumed default value of 0.25
L6	Molecular weight of waste	18 for unit concentration run; 147 for pure component run
L7	For aqueous waste, enter 1	1 for unit concentration run; 0 for pure component run
L8	Time of calculation (d)	Dependent on type of WMU
L9	For biodegradation, enter 1	Dependent on type of WMU
L10	Temperature (°C)	Set by location of WMU
L11	Windspeed (m/s)	Set by location of WMU
L12	Area (m^2)	Input from Ind D Survey
L13	Fraction organic carbon	Assigned randomly from distribution

meteorological data input that potentially impacts the emission results for the CHEMDAT8 model for the land-based WMU.

The total porosity and air porosity values that were used in the emission assessments were the default CHEMDAT8 model values for these parameters. These assumed porosity values appear to be reasonable for a waste and waste/soil matrices that have a density of 1.1 g/cm^3.

For aqueous wastes, the molecular weight of the waste (Input ID No. L6) does not impact the calculations.

The molecular weight of the waste for the "pure component" runs using Raoult's law was set to 147 g/mol, which is the CHEMDAT8 default value for this input parameter. If the waste were truly pure constituent, then the appropriate molecular weight input for the waste would be the specific constituent's molecular weight. However, the pure component run is used to backcalculate an appropriate waste concentration limit that is often considerably less than pure component. Therefore, the scenario modeled is not actually pure constituent, and modeling the waste at the molecular weight of the constituent is not appropriate. If the actual molecular weight of the waste is higher than 147 g/mol, the default molecular weight used may underestimate volatile emissions. Conversely, if the actual molecular weight of the waste is lower than 147, the default value may overestimate volatile emissions. The magnitude of this under- or over estimation is expected to be small over the range of likely waste molecular weights.

The process of biodegradation is an important one because it lowers both the emission rate and the average soil concentration. Consequently, biodegradation is an important input parameter, and the biodegradation rate constants used in the model are critical parameters.

71

Biodegradation was treated differently for the various WMUs. Landfills are not designed for biodegradation, and waste in wastepiles managed over short periods will not be affected substantially. Therefore, both the landfill emission runs and the short-term wastepile emission runs did not include biodegradation losses. First-order biodegradation was included in the LAU emission runs and long-term wastepile emission runs. Note that the default CHEMDAT8 model method of calculating biodegradation rates was not used. CHEMDAT8 biodegradation rates were derived primarily from wastewater studies and applied to model biodegradation in soils using an assumed, low-biomass concentration. Because the first-order biodegradation rate constants obtained from Howard et al. (1991) were either based on soil studies or explicitly evaluated for applicability to soil, they were used instead. These biodegradation rate constants provide a more direct link to soil-based biodegradation and are considered more appropriate for modeling biodegradation in land-based WMUs.

The fraction of organic carbon, f_{oc}, affects adsorption. The f_{oc} of interest is the f_{oc} of the waste (assuming the waste contains sludge/solids) and not of the soil. Little data exist concerning the f_{oc} of the waste itself; therefore, default values for this parameter were defined and applied in the absence of data. A distribution was developed to represent f_{oc} at all sites. Because this parameter is a fraction, it must range from 0 to 1. A beta distribution was selected for the distributional form because the beta distribution also varies between 0 and 1. This distribution is defined by two parameters called alpha and beta. Fraction organic carbon is waste- and site-specific but is most often less than 0.1. Therefore, the distribution was fitted with the criteria that half the values generated should be less than 0.1, and that 90 percent of the values should be less than 0.5. The fitted distribution has an alpha value of 0.455 and a beta value of 2.05. Individual f_{oc} values for each WMU were then selected randomly from the distribution.

4.2.2.2 Particulate Emissions. Particulate emissions due to wind erosion were modeled for land-based units (landfills, land application units, and wastepiles). Landfills and LAUs were modeled differently than wastepiles because they are ground-level sources and wastepiles are elevated sources. Wind erosion emissions from landfills and LAUs were modeled using the Cowherd model (U.S. EPA, 1985b, 1988). This model estimates the emission of respirable particles (i.e., PM_{10}) due to wind erosion from a ground-level surface with an unlimited reservoir of erodible particles. Surfaces are defined as having a limited or unlimited reservoir based on threshold friction velocity (U^*); surfaces with a U^* greater than 0.5 m/s are considered limited; those with U^* less than 0.5 m/s are considered unlimited (U.S. EPA, 1988). Threshold friction velocity is a measure of the windspeed at the ground surface that would be required to remove particles from the surface. Examples of limited reservoirs include nonhomogeneous surfaces with stones, clumps of vegetation, or other nonerodible elements or crusted surfaces. Further, wind erosion is considered unlikely to occur from surfaces with full vegetative cover.

Wind erosion emissions from wastepiles were modeled using an equation from AP-42 (U.S. EPA, 1985a) for estimating emissions from wind erosion from active storage piles. The equation gives emissions of total suspended particulates (TSP). Typically, an equation-specific particle size multiplier would be applied to reduce the emissions to a desired size category, in this case, PM_{10}. No particle size multipliers (PSM) are given for this equation in AP-42; however, Cowherd (U.S. EPA, 1988) gives a PM_{10} particle size multiplier of 0.5 for use with this equation.

72

Important input parameters for particulate emissions include silt content of waste (i.e., percent with small particle size), number of days with greater than 0.01 inches of rainfall, and percent of time that windspeed exceeds 5.4 m/s. Data on the silt content of the wastes being modeled were not available. A median silt content for miscellaneous fill material of 12 percent (U.S. EPA, 1988) was used. The number of precipitation days and the frequency of windspeed greater than 5.4 m/s were location-specific; values were obtained from the National Oceanic and Atmospheric Administration (NOAA, 1992) and are summarized in Section 4.7.2.

4.2.3 Critical Parameters for Tank Emissions Model

Table 4-2 presents the required CHEMDAT8 input parameters[2] along with units and comments on the source of the parameter values. As shown in Table 4-2, only one parameter (flow rate) has values that are taken directly from the TSDR survey data as discussed in Section 3.4 (that is, the data were provided by facility owner/operators at the time the survey was conducted). Volume data, which were used to impute a number of model input parameter values for tanks, were also taken directly from the TSDR survey data. The imputation procedures for the aeration and waste characteristics parameter values are discussed in this section. The procedures applied to estimate unit design parameters, which are also critical to air dispersion modeling, are discussed in Section 3.4.

Factors that affect the relative surface area of turbulence and the intensity of that turbulence are important in determining the fate of chemicals in tanks. The tank model has several input parameters that impact the degree and intensity of the turbulence created by the aeration (or mixing). Note that many of these parameters are determined for both aerated and some nonaerated (both treatment and storage) tanks. For nonaerated tanks with an above-surface intake (based on information available from the survey data), a small degree of aeration is modeled to account for the agitation of splash loading. Although there is not actually an aerator in these tanks, one is characterized to simulate the effects of splash loading. For nonaerated tanks with below-surface intakes, no agitation is modeled, and the aeration parameters are all set to zero. The values of most of the aeration parameters were estimated based on data collected in 1985 and 1986 during EPA site visits to aerated treatment systems (Coburn et al., 1988; Eichinger, 1985) and hypothetical tanks from the TSDF BID (U.S. EPA, 1991) (see Section 3.4.2 for further details). Aerator parameters for 16 tanks and surface impoundments (which are expected to have similar aerator properties to tanks) from these site visits and the TSDF BID are shown in Table 4-3. These data were used to provide a sense of the range and typical values of the aeration parameters.

The tank model is most sensitive to the fraction aerated; the total power, number of aerators, and impeller diameter have some impact on the emission results; and the other parameters have little to no impact on the estimated emissions.

[2] Note that this table also includes one parameter, height, that is not used in CHEMDAT8 but is used in dispersion modeling. For clarity, it is presented here as it is derived from depth.

Table 4-2. CHEMDAT8 Tank Model Input Requirements

Input Parameter	Source	Imputation Method
Unit Design		
Flow rate (m^3/s)	Survey	None
Depth (m)	Imputed	Based on volume
Average surface area (m^2)	Imputed	Based on volume and depth
Height above ground (m)	Imputed	Based on depth
Aeration Parameters		
Fraction agitated	Estimated distribution	
Total power (hp)	Imputed	Based on volume
Number of impellers	Imputed	Based on total power
Impeller diameter (cm)	Estimated constant = 61	
Impeller speed (rad/s)	Estimated constant = 130	
Power efficiency (unitless)	Estimated constant = 0.83	
O_2 transfer rate (lbO^2/h-HP)	Estimated constant = 3	
Submerged air flow (m^3/s)	Estimated constant = 0	
Waste Characteristics		
Active biomass conc. (kg/m^3)	Estimated distribution	Depends on treatment code
Total solids in (kg/m^3)	Estimated distribution	
Total organics (COD) In (g/m^3)	Estimated distribution	
Total biorate (mg/g-h)	Estimated constant = 19	
Meteorological Data		
Temp (°C)	Imputed	Based on meteorological station
Windspeed (m/s)	Imputed	Based on meteorological station

The fraction aerated depends on the level of aeration. Distributions for this parameter were developed for each aeration level. Highly aerated tanks should have a higher fraction aerated than less-aerated tanks. No tank can have a fraction aerated greater than 1 or less than zero, and realistically, the fraction aerated for an aerated tank should not be close to zero. A non aerated tank may have a small fraction aerated to simulate the agitation from splash loading from an above-surface intake. For HI aeration tanks, fraction aerated is randomly assigned from a normal distribution with a mean of 0.75 and a standard deviation of 0.1. Values greater than 1 are truncated to 1. For LO aeration tanks, fraction aerated is randomly assigned from a uniform distribution with endpoints of 0.2 and 0.8. For NO aeration and storage tanks with above-surface intakes, fraction aerated was randomly assigned from a normal distribution with a mean of 0.08 and a standard deviation of 0.03. Values less than zero were truncated at zero, and values that implied an area agitated more than 10 m^2 were truncated so that the agitated area was 10 m^2. The rationale for this is that splash loading would not affect an area greater than 10 m^2. For NO aeration tanks with below-surface intakes, the fraction aerated was set to zero.

74

Table 4-3. Summary of Mechanical Aerator Information Collected in EPA Site Visits for RCRA Air Emission Standards

Type of unit	Aeration	Total Power (hp)	# Aerators	Impeller diameter (cm)	Impeller speed (rad/s)	Oxygen transfer rate (lb/hp-h)	Power/ volume (hp/m^3)	Aerator Power for Total Power >100 hp (hp)
Aeration tank	HI	300	3				0.053	100
Aeration tank	HI	7.5	1				0.067	
Aeration tank	HI	900	9	259	188	3.0	0.022	100
Aux. acr. tank	HI	450	6	259	126	3.0	0.021	75
Aeration tank	HI	900	6	274			0.034	150
Aeration tank	HI	150	2	50	124	3.0	0.045	75
Aerated trtmnt tank	HI	7.5					0.069	
Aerated trtmnt tank	HI	120					0.075	
So eq. basin	LO	30	2	107	7.1		0.125	
No eq. basin	LO	20	1	152	5.9		0.105	
Mixing tank	LO	3	1				0.027	
Eq. basin	LO	150	5	122	7.1		0.004	30
Mixing tank	LO	1.5	1	183	1.0		0.022	
Aerated lagoon	SI	30	2	14	367		0.009	
Aerated lagoon	SI	270	6	42	123		0.011	45
Aerated lagoon	SI	1800	28				0.040	64

Total aerator power depends on the volume of the tank and the level of aeration. For HI aeration tanks, total power per million gallons of volume was randomly assigned using a normal distribution with a mean of 115 hp/million gallons and 90 percent of the values between 80 and 150 hp/million gallons (Metcalf and Eddy, 1979). For LO aeration tanks, total power per million gallons of volume was randomly assigned using a normal distribution with a mean of 30 hp/million gallons and 90 percent of the values between 15 and 45 hp/million gallons This was based on industry comments on the 1998 ACS that indicated that surface aerators used for mixing typically have power levels between 15 and 20 hp/million gallons, and aerators used for activated sludge have a minimum power of 20 to 30 hp/million gallons. The upper value of 30 was adjusted upward by a factor of 1.5 to provide values above the minimum, and the resulting range of 15 to 45 hp/million gallons was presumed to encompass 90 percent of all values. For both HI and LO aerated tanks, the power per million gallons was multiplied by the volume in millions of gallons to determine total power. For NO aeration and storage tanks, the power per million gallons was determined in the same way as for LO aeration tanks. However, the total power was then calculated by multiplying the power per million gallons by fraction aerated and volume (in million gallons) to estimate total power. A minimum value of 0.25 hp was set. Fraction aerated is included in this calculation to account for the fact that the whole tank volume is not affected by the agitation caused by splash loading.

The number of aerators (or impellers) for aerated tanks was derived from the total power and the power per aerator. Typical values of power per aerator for different total power levels were based on the data in Table 4-3 and the number of aerators set as follows:

75

For Total Power ≤ 25 HP, one aerator

For 25 HP < Total Power < 80 HP, randomly pick one or two aerators with equal probability

For Total Power ≥ 80 HP, randomly pick a power per aerator using a uniform distribution with endpoints of 60 and 100, then divide the Total Power by the random power per aerator and round up to the next integer. While some of the power per aerator values associated with the data in Table 4-3 fell outside the range of 60 to 100, this range represented most of the values.

For nonaerated tanks, the number of impellers was set to 1 if the tank had an above-surface intake and to zero if it had a below-surface intake.

Impeller diameter and rotational speeds appeared to be related parameters. Generally, the longer (or high-diameter) impellers found in the site visit data (see Table 4-3) had lower rotational speeds and the shorter impellers had faster rotational speeds. Consequently, independent random assignments of these variables was determined to be inappropriate. Rather than attempting to develop a correlation between the two parameters based on limited data, the fixed values used for the model tanks developed for the Hazardous Waste TSDF air rules (U.S. EPA, 1991) were selected. These values are reasonable central tendency values based on the limited available data presented in Table 4-3:

Impeller diameter fixed at 61 cm (2 ft), based on data reported by Watkins (1990)

Impeller rotational speed fixed at 130 rad/s, based on data reported by Watkins (1990)

Oxygen transfer rating and power efficiency do not have much impact on the emission, and vary over a very small range. Therefore, these values were fixed, as follows:

Oxygen transfer rating fixed at 3.0 lb O_2/HP-h; U.S. EPA (1991) reports a range of 2.9 to 3 lb O_2/hp-h

Power efficiency fixed at 0.83; U.S. EPA (1991) reports a range of 0.8 to 0.85.

Submerged air flow was set to zero for all tanks because aeration is modeled as mechanical aeration, not diffused air.

Waste characteristics that influence the rate of biodegradation are important in determining emissions from both aerated and storage tanks. As shown in Table 4-2, these parameters include active biomass concentration, total solids in, total organics in, and total biorate. Limited data were available on waste characteristics for tanks; most facilities only need to measure these parameters for the final wastewater discharge and do not measure them in the influent. From a review of the site visit reports described previously, the data estimations

76

discussed below were derived and used for aerated tanks reporting biological treatment. Aerated tanks reporting other types of treatment, nonaerated treatment tanks, and storage tanks were modeled with no biodegradation.

Unlike the biodegradation rate model that was used for the land-based units, the biodegradation rate model used in CHEMDAT8 for tanks is dependent on the amount of active biomass in the WMU. Therefore, the active biomass concentration is a critical parameter for aerated tanks. Because this parameter can vary widely for different types of tanks, biomass concentrations were set on a tank-by-tank basis for aerated tanks using process code information (WMU codes) from the TSDR Survey. For biological treatment aerated tanks, the active biomass concentration measured as mixed liquor volatile suspended solids (MLVSS) from the site visit reports range up to 4 g/L, and one source test measured MLVSS up to 6 g/L. However, the typically observed MLVSS concentration fell in the 1.5- to 3.0-g/L range. Many of the biodegradation rate constants developed for CHEMDAT8 used 2 g/L as a default MLVSS concentration to normalize the constituent disappearance rates. Therefore, 2 g/L was considered the most appropriate central tendency value for the active biomass concentration. Consequently, the following algorithm was used to select the active biomass concentration.

- For biological treatment units (WMU codes 52WT and 58WT), this value was randomly assigned for each tank, using a uniform distribution with endpoints of 1 and 3 g/L

- For all other aerated tanks (as well as storage and nonaerated treatment tanks), active biomass concentration was set to 0 g/L.

The "total biomass(solids) in," "total organics in," and "total biorate" (Input ID Nos. T6, T8, and T9) impact the rate of biomass production and subsequently the amount of contaminant that is absorbed onto the solids. (Note: The "biomass solids in" does not affect the biodegradation rate and is more appropriately labeled simply "solids in."). These inputs, however, have little or no impact on the estimated emission rates for most of the contaminants modeled in this analysis. The value for "total solids in" was randomly assigned for each tank using a uniform distribution with endpoints of 0.1 and 1 g/L, based on best professional judgment. CHEMDAT8 (U.S. EPA, 1994e) suggests a range of 0.1 to 0.4 for surface impoundments designed for biodegradation. The "total organics in" value was randomly assigned for each tank, using a uniform distribution with endpoints of 100 and 1,000 mg/L, based on best professional judgment. The CHEMDAT8 default value is 250 mg/L (U.S. EPA, 1994e).

The input parameter listed as "total biorate, mg/g_{bio}-h" is used in conjunction with the active biomass concentration to estimate the growth or replacement rate of biomass, e.g., how much of the "old" biomass is consumed to grow "new" biomass that is now available for contaminant adsorption. This biomass replacement rate is used in conjunction with the influent total solids and total organic concentrations to determine the total rate at which total suspended solids (TSS) are removed (or "wasted") from the system. Because this input parameter impacts only tanks that have active biomass greater than zero (a small fraction of the total number of tanks in the data set) and this biomass replacement factor will generally have only a small, if any, contribution to the adsorptive losses, no additional research was performed for this parameter.

77

Instead, the total biorate is fixed at 19 mg/g biomass-h, which is the default value recommended for CHEMDAT8 (U.S. EPA, 1994e).

Due to the nonlinearity of the biodegradation rate model used in the tank emission estimates, direct backcalculation of an acceptable waste concentration may not be appropriate for some compounds. Unlike the emission results from the land-based units, the contaminant concentration used in the analysis may impact the predicted "normalized" emission rate (i.e., the emission rate in g/m^2-s per mg/L of contaminant). Therefore, the tanks with biodegradation were run at a low concentration (i.e., 0.001 mg/L) and at a high concentration (i.e., the constituent's solubility). The most appropriate backcalculated emission value was then selected based on the concentration range of the backcalculated values and the constituent's biodegradation characteristics (see Section 7.9 for further details).

Meteorological inputs are also important for the tank emission model. For nonaerated treatment tanks and storage tanks, the emission estimates are impacted by both temperature and windspeed. Because the emissions for aerated tanks are predominantly driven by the turbulent area and associated mass transfer coefficients, the emissions from the aerated tanks are not strongly impacted by the windspeed. Aerated tank emissions are impacted by temperature. Annual average temperatures were used as input to the model based on tank locations. (Note that, dependent on the residence time of the waste in the tank, the temperature of the waste in the tank was not expected to vary significantly with changing atmospheric temperatures, and annual average temperatures were used to estimate the average waste temperature in the tanks). The location of each tank is available from the TSDR survey data. Based on this information, each tank was assigned to one of the 29 meteorological stations used in the dispersion modeling. As discussed in Section 5, these assignments were made based on both proximity and similarity of the climatological characteristics that affect meteorological data. The windspeed and temperature used in emissions modeling are the annual averages for the assigned meteorological station, taken from NOAA (1992).

4.3 Modifications to CHEMDAT8 for Land-Based Units

The CHEMDAT8 model estimates emissions from land-based WMUs (such as land application units, open landfills, and wastepiles) using a simple emissions model that accounts for contaminant partitioning between a liquid waste matrix and the air, diffusion of vapors through a porous media, and contaminant loss through biodegradation. The CHEMDAT8 model, however, does not accommodate entered total waste concentrations (i.e., liquid and solid phase). The assumption of an entered waste concentration in liquid phase was based on the petroleum wastes for which CHEMDAT8 was originally developed and may not apply to the chemicals considered in this analysis. Therefore, a method for including adsorptive partitioning for total waste concentrations was developed and is presented below.

Assuming three-phase partitioning (adsorbed, dissolved, and volatile), the total concentration of a contaminant can be expressed as the sum of the masses of the contaminant adsorbed on the soil, dissolved in the liquid, and in the air spaces divided by the total mass of contaminated soil as follows:

78

$$C_T = C_s \rho_b + \theta_w C_w + \theta_a C_a \quad (4\text{-}1)$$

where

C_T = total contaminant concentration ($\mu g/cm^3_{soil} = g/m^3_{soil}$)
C_s = concentration of contaminant adsorbed on soil ($\mu g/g_{soil} = g/Mg_{soil}$)
ρ_b = soil dry bulk density ($g_{soil}/cm^3_{soil} = Mg_{soil}/m^3_{soil}$)
θ_w = water-filled soil porosity (m^3_{water}/m^3_{soil})
C_w = concentration of contaminant dissolved in liquid ($\mu g/cm^3_{water} = g/m^3_{water}$)
θ_a = air-filled soil porosity (m^3_{air}/m^3_{soil})
C_a = concentration of contaminant in air ($\mu g/cm^3_{air} = g/m^3_{air}$).

The adsorbed contaminant concentration is assumed to be linearly related to the liquid phase concentration as follows:

$$C_s = K_d C_w \quad (4\text{-}2)$$

where

C_s = concentration of contaminant adsorbed on soil ($\mu g/g_{soil} = g/Mg_{soil}$)
K_d = soil-water partition coefficient ($cm^3/g = m^3/Mg$)
C_w = concentration of contaminant dissolved in liquid ($\mu g/cm^3_{water} = g/m^3_{water}$)

For organic constituents:

$$K_d = K_{oc} f_{oc} \quad (4\text{-}3)$$

where

K_{oc} = organic-carbon partition coefficient ($cm^3/g = m^3/Mg$)
f_{oc} = weight fraction organic carbon content of the solid matrix ($g/g = Mg/Mg$).

The contaminant concentration in the vapor phase is assumed to be linearly related to the liquid phase concentration as follows:

$$C_a = H' C_w \quad or \quad C_w = \frac{C_a}{H'} \quad (4\text{-}4)$$

where

H' = dimensionless Henry's law constant = H/RT = 41 × H at 25 °C where H = Henry's law constant at 25 °C ($atm\text{-}m^3/mol$).

79

Combining Equations 4-1, 4-2, and 4-4 by replacing C_s in Equation 4-1 with the term in Equation 4-2 and C_w in Equation 4-1 with the term in Equation 4-4 yields the following expression in terms of the gas phase concentration C_a:

$$C_T = C_a \left[\frac{\rho_b K_d}{H'} + \frac{\theta_w}{H'} + \theta_a \right] \quad (4\text{-}5)$$

The total contaminant concentration, C_T, represents the measured soil concentration. Equation 4-5 can be rearranged to calculate the gas phase concentration given the total contaminant concentration as follows:

$$C_a = \frac{C_T H'}{K_d \rho_b + \theta_w + \theta_a H'} \quad (4\text{-}6)$$

This partitioning theory, as represented by the above equations, was used to include adsorption in CHEMDAT8, as described below.

The CHEMDAT8 land treatment emission model is based on the diffusion of a gas from a semi-infinite slab that initially has a uniform concentration of diffusing material throughout and that has equal concentrations of diffusing material at each surface (U.S. EPA, 1994e). The emission equations presented in CHEMDAT8 are in terms of the mass (as opposed to concentration) of contaminant in the gas phase.

CHEMDAT8 uses an equilibrium partitioning factor, K_{eq}, as a multiplier to correct the effective diffusion coefficient. The partitioning factor, K_{eq}, represents the ratio of the mass of organics in the vapor phase to the mass of organics in the soil/waste mixture and, therefore, is used to estimate the amount of material that partitions into the vapor phase based on equilibrium conditions within the soil/waste mixture. The CHEMDAT8 solution, as described in the CHEMDAT8 documentation, requires a ratio of the total mass of contaminant in the gas phase to the total mass of contaminant in the soil/waste matrix. The mass ratio for the partitioning correction factor (including adsorption) that was used in the CHEMDAT8 model, $K_{eq,\ ads}$, therefore, was defined as follows:

$$K_{eq,\ ads} = \frac{M_a}{M_T} \quad (4\text{-}7)$$

where

M_a = mass of constituent in the air-filled soil porosity (g)
M_T = total mass of constituent in the soil/waste mixture (g).

40

The masses of constituent in the air-filled soil porosity and in the soil/waste mixture are equal to their respective concentrations times volumes:

$$\frac{M_a}{M_T} = \frac{C_a V_a}{C_T V_T} \quad (4\text{-}8)$$

where

V_a = total volume in the air-filled soil porosity (m^3)
V_T = total volume of the soil/waste mixture (m^3).

Using the relationship between C_a and C_T presented in Equation 4-6, and the definition of porosity shown in the following equation,

$$\theta_a = \frac{V_a}{V_T} \quad , \quad (4\text{-}9)$$

the terms in Equation 4-8 can be substituted for, and $K_{eq, ads}$ can be rewritten as follows:

$$K_{eq, ads} = \frac{\theta_a H'}{(\rho_b K_d + \theta_w + \theta_a H')} \quad (4\text{-}10)$$

The CHEMDAT8 model was modified to include Equation 4-10. This equation presents an expression for $K_{eq, ads}$ that achieves the goal of including adsorptive partitioning of total waste concentration in the model. This equation always yields a partitioning value of 1 or less. At high Henry's law values, $K_{eq, ads}$ is necessarily equal to 1, providing the same emission rate predictions as if the total initial mass of contaminant was in the vapor phase with no partitioning.

4.4 Development of Volatile Emissions and Waste Concentrations for Landfills

The basic assumptions used for modeling landfills are as follows:

- The landfill operates for 20 years filling 20 cells of equal size sequentially.
- The active cell is modeled as being instantaneously filled at time t=0 and remains open for 1 year.
- Emissions are calculated only for one cell for 1 year (after 1 year, the cells are either depleted of the constituent or capped).
- The waste is homogeneous with an initial concentration of 1 mg/kg.

81

- The waste matrix may be aqueous (Henry's law partitioning applies) or organic (Raoult's law partitioning applies).

- Annual average temperature is used (determined by assigned meteorological station).

- Acute and subchronic exposures were not modeled.

The results presented in Volumes I and III are based on the aqueous-phase emission rates (i.e., assuming a concentration of 1 mg/kg and Henry's law partitioning). Most of the waste streams managed in land-based units are expected to contain constituents in the aqueous, rather than the organic, phase; therefore, this is the most realistic scenario. However, results based on organic phase emissions are of interest in two circumstances:

- Most chemicals are better able to volatilize from an aqueous medium than from an organic medium; therefore, for most chemicals, the aqueous-phase emission rates are considerably higher than organic-phase emission rates. However, for a few chemicals (most notably formaldehyde), the organic-phase emissions are higher than the aqueous-phase emissions. When this is the case, the results based on aqueous-phase emissions are footnoted to indicate that organic-phase emissions would be higher, so a concentration based on organic-phase emissions would be lower. This does not invalidate the aqueous-phase results, as that is still the most likely waste matrix.

- Some of the backcalculated waste concentrations based on aqueous-phase emissions exceed the soil saturation concentration at a neutral pH and temperature of 20 to 25°C. This is the maximum possible aqueous-phase concentration in soil; once this is exceeded, free (organic-phase) product will occur in the soil. This soil saturation concentration has been estimated for each chemical in the analysis, but this is a somewhat site- and waste-specific value because it depends on solubility, soil properties such as bulk density and porosity, and temperature (see Section 7.10.2 for the equation used to calculate the soil saturation concentration). Therefore, a backcalculated concentration may exceed it in some situations but not in others. When the backcalculated concentration based on aqueous-phase emissions exceeded the calculated soil saturation concentration at a neutral pH and temperature of 20 to 25°C, the result was footnoted to indicate whether pure component (i.e., a concentration of 10^6 mg/kg) would result in a risk exceeding the target risk when modeled using organic phase assumptions and Raoult's law.

Table 4-1 provides the CHEMDAT8 model input requirements for land-based units with some commentary about each input parameter. The inputs that were calculated from the Industrial D Screening Survey data were calculated as follows:

82

All total quantities, capacities, and areas in the Industrial D Screening Survey were divided by the number of landfills at the facility to get landfill-specific estimates.

Loading = bulk density = 1.1 g/cm^3.

Tilling depth (cm) = landfill depth, l, calculated as follows:

$$D_{till} = \frac{100 \times C}{A \times BD} \qquad (4\text{-}11)$$

where

D_{till} = landfill depth (cm)
100 = unit conversion (cm/m)
C = capacity (Mg)
A = landfill area (m^2)
BD = bulk density (g/cm^3 = Mg/m^3)

If the calculated depth was less than 2 feet or more than 33 feet, then the method described in Section 3.1 was used.

Total landfill surface area was divided by 20 to get surface area of landfill cell.

The total landfill capacity was divided by 20 to get the average annual quantity of waste, Q_{annual}.

The landfill cell areas and depth were entered into the CHEMDAT8 input table (along with average ambient temperature), and the emission fraction for the "intermediate time" (365.25 days) was calculated. This emission fraction was then multiplied by the annual waste quantity and waste concentration and divided by the area of the cell to calculate the output emission rate as follows:

$$E = \frac{Q_{annual} \times C_{waste} \times f_{emit}}{A_{cell} \times 31{,}557{,}600} \qquad (4\text{-}12)$$

where

E = emission rate (g/m^2-s)
Q_{annual} = annual waste quantity (Mg/yr)
C_{waste} = waste concentration (mg/kg = g/Mg)
f_{emit} = fraction emitted (unitless)
A_{cell} = cell area (m^2)
31,557,600 = unit conversion factor (s/yr).

The average concentration of the waste in the landfill cell was estimated from the emission fraction and the biodegradation fraction (although the biodegradation fraction was zero – no biodegradation – for the landfill) by assuming first-order contaminant (concentration) disappearance. Assuming first-order kinetics with respect to the contaminant concentration in the landfill cell, an exponential decay can be written in terms of the apparent overall first-order decay rate. The concentration at a given time is equal to the initial concentration as follows:

$$C_{waste,t} = C_{waste,o} \times e^{(-K1,all\ t)} \quad (4\text{-}13)$$

where

$C_{waste,\ t}$ = waste concentration at time t (mg/kg)
$C_{waste,\ 0}$ = waste concentration at time 0 (mg/kg)
$K_{1,\ all}$ = apparent first-order decay rate (yr^{-1})
t = time period of calculation (yr).

At the end of 1 year, $C_{waste,t}/C_{waste,o}$ = 1 - emission fraction - biodegraded fraction. Therefore, the $K_{1,all}$ t term, at the time period for which the fraction loss terms were calculated, is simply:

$$K_{1,all}\ t = -\ln\ (1 - f_{emit} - f_{bio}) \quad (4\text{-}14)$$

where

$K_{i,\ all}$ = apparent first-order decay rate (yr^{-1})
t = time period of calculation (yr)
f_{emit} = fraction emitted (unitless)
f_{bio} = fraction biodegraded (unitless).

The concentration versus time profile (Equation 4-13) can then be integrated to calculate the average waste concentration, $C_{waste,ave}$, over the time period of the calculation:

$$C_{waste,ave} = C_{waste,0} \times \frac{1 - e^{-K_{1,all}\ t}}{K_{1,all}\ t} \quad (4\text{-}15)$$

where

$C_{waste,\ ave}$ = average waste concentration (mg/kg)
$C_{waste,\ 0}$ = waste concentration at time 0 (mg/kg)
$K_{1,\ all}$ = apparent first-order decay rate (yr^{-1})
t = time period of calculation (yr).

The input parameters required for the landfill are presented in Table 4-1. The annual waste quantity and unit dimensions are the critical source parameters. For landfills, the loading

84

rate is pure waste material so that loading (Input ID No. L1) is basically the waste density. A waste density of 1.1 g/cm^3 was used for the landfill to be consistent with the waste densities used in the analysis of the Industrial D Screening survey data. The annual waste quantity is also combined with the area of the landfill to calculate the depth of the landfill.

Temperature and porosities have some impact on predicted emissions. The biodegradation flag was set to zero (no biodegradation) for landfills. Therefore, temperature variations should have less of an impact on the annual emission rates from landfills than from land application units. The model is insensitive to molecular weight of the waste (for aqueous wastes) and windspeed (for long-term emission estimates).

4.5 Development of Volatile Emissions and Waste Concentrations for Land Application Units

4.5.1 Chronic Exposure Analysis

Because the same basic CHEMDAT8 model was used for landfills and land application units, the emission estimates for land application units have some similarities to the landfill emission estimates, but there are also a number of differences. The basic modeling assumptions used for modeling land application units are as follows:

- # The land application unit emissions are modeled as pseudo-steady-state. Emissions are actually time-dependent (depending on how recently waste has been added) but are modeled as a series of steady-state emissions for short time intervals, which are then averaged to produce a long-term emission rate.

- # Emissions in year 40 are used to estimate long-term emissions. This does not reflect an assumed operating life of the unit but is simply a sufficiently long period to ensure that steady state has been reached, if it is ever going to be, (typically, steady state is reached in 1 or 2 years) and to exceed most of the exposure durations used in the modeling.

- # Waste application occurs twice monthly (i.e., 24 times per year).

- # The waste is homogeneous with an initial concentration of 1 mg/kg.

- # The waste matrix may be aqueous (Henry's law partitioning applies) or organic (Raoult's law partitioning applies).

- # Temperature is determined by assigned meteorological station; monthly average temperature was used.

- # Biodegradation occurs at temperatures greater than 5 °C.

The results presented in Volumes I and III are based on the aqueous-phase emission rates (i.e., assuming a concentration of 1 mg/kg and Henry's law partitioning). Most of the waste

streams managed in land-based units are expected to contain constituents in the aqueous, rather than the organic, phase; therefore, this is the most realistic scenario. However, results based on organic phase emissions are of interest in two circumstances:

Most chemicals are better able to volatilize from an aqueous medium than from an organic medium; therefore, for most chemicals, the aqueous-phase emission rates are considerably higher than organic-phase emission rates. However, for a few chemicals (most notably formaldehyde), the organic-phase emissions are higher than the aqueous-phase emissions. When this is the case, the results based on aqueous-phase emissions are footnoted to indicate that organic-phase emissions would be higher, so a concentration based on organic-phase emissions would be lower. This does not invalidate the aqueous-phase results, as that is still the most likely waste matrix.

Some of the backcalculated waste concentrations based on aqueous-phase emissions exceed the soil saturation concentration at a neutral pH and temperature of 20 to 25 °C. This is the maximum possible aqueous-phase concentration in soil; once this is exceeded, free (organic-phase) product will occur in the soil. This soil saturation concentration has been estimated for each chemical in the analysis, but this is a somewhat site- and waste-specific value because it depends on solubility, soil properties such as bulk density and porosity, and temperature (see Section 7.10.2 for the equation used to calculate the soil saturation concentration). Therefore, a backcalculated concentration may exceed it in some situations but not in others. When the backcalculated concentration based on aqueous-phase emissions exceeded the calculated soil saturation concentration at a neutral pH and temperature of 20 to 25 °C, the result was footnoted to indicate whether pure component (i.e., a concentration of 10^6 mg/kg) would result in a risk exceeding the target risk when modeled using organic phase assumptions and Raoult's law.

The inputs that were calculated for the land application units were calculated as follows:

The total annual waste quantities and surface areas for each facility, as reported in the Industrial D Screening Survey, were divided by the number of LAUs at the facility to get LAU-specific estimates.

Tilling depth (cm) = 20 cm if Q_{annual} (Mg/yr)/Area(m^2) ≤ 0.2 . If Q_{annual}/Area > 0.2, then depth (cm) = 100 × Q_{annual}/Area.

Loading rate, L, is calculated as follows:

$$L = \frac{Q_{annual} \times t}{A \times D_{till}} \qquad (4\text{-}16)$$

86

where

L = loading rate ($Mg/m^3 = g/cm^3$)
Q_{annual} = annual waste quantity (Mg/yr)
t = time period of calculation (yr)
A = LAU area (m^2)
D_{till} = tilling depth (m).

Biodegradation is assumed to occur if temperature is greater than 5°C. If the temperature is 5°C or lower, biodegradation is turned off.

Time of calculation = 365.25/24 applications per year = 15.2 days.

Monthly temperature and windspeeds were calculated by averaging the hourly temperature and windspeeds.

A sensitivity analysis was conducted that investigated the impact that application frequency (monthly versus quarterly) and the averaging period for meteorological data (monthly versus yearly) had on emission estimates. Results from this analysis indicated that both application frequency and averaging period do impact emissions. Therefore, modeling was conducted using monthly average meteorological data. In the absence of reported unit-specific data on application frequency in the Industrial D database, it was assumed that waste was applied to the unit two times during each month of the year (i.e., 24 times per year).

Land treatment generally involves the application of wastes to the land in either a liquid or a semi-solid form with treatment occurring through the biological degradation of the hazardous constituents. Waste is assumed to be delivered by tank trucks and is applied uniformly across the entire unit area. Based on the literature reviewed, it appears that the frequency at which waste is to be applied is dependent on a number of variables, including waste characteristics (e.g., constituent concentrations and oil content), soil type, vegetation, and climatic conditions. The application frequencies found in literature range from yearly to an extreme of 260 times per year. U.S. EPA (1989), which cites data presented in Land Treatment Practices in the Petroleum Industry (*Environmental Research & Technology*, 1983), reports a typical range for refineries of 2 to 52 applications per year. Martin et al. (1986) presents frequencies for 13 operating petroleum refineries that represent the geographical distribution of refineries. The reported frequencies ranged from 1 to 260 times per year, with the majority of the sites reporting monthly or yearly frequencies. The *Handbook of Land Treatment Systems for Industrial and Municipal Wastes* (Reed and Crites, 1984) reports that an application of once per week is commonly used but suggests that determination of application frequency should consider site-specific conditions. One of the site-specific variables that can impact the frequency of application is the number of months out of the year the unit is active. Martin et al. (1986) reported that the number of months facilities were actively used varied from 6 months (colder climates) to 12 months (warmer climates).

Based on the information from the literature review described above, the relationship between number of applications per year and waste quantity shown in Table 4-4 was developed.

The relationships shown in Table 4-4 were applied to the Industrial D data set for land application units. Table 4-5 summarizes the frequency of waste application estimated using the Industrial D data. As shown, approximately 86 percent of the land application units were predicted to have application frequencies 52 times or less per year and only 26 percent were estimated to have waste applied 4 times or less per year. The median number of applications per year is 16.

It was not feasible from a modeling perspective to assign different frequencies of application to each unit. Also, the use of monthly meteorological data suggested an application frequency that could be expressed as an integer on an applications per month basis. Therefore, an application frequency of 24 applications per year (2 applications per month) was selected as best representing the data available within these modeling constraints.

Table 4-4. Relationship Between Frequency of Application and Waste Quantity

Annual Waste Quantity at Unit (Mg/yr)	Number of applications/year
≤1,500	Annual waste quantity at unit/15
≥1,500, <15,000	Annual waste quantity at unit/150
≥15,000, <150,000	Annual waste quantity at unit/1,500
≥150,000	Annual waste quantity at unit/15,000

Table 4-5. Estimated Frequency of Application Using Industrial D Data

Frequency of Waste Application	Total Number of Land Application Units	Percentage of Total Land Application Units
≤1/yr	49	14
≤4/yr	90	26
≤12/yr	133	39
≤16/yr	176	51
≤24/yr	217	62
≤52/yr	295	86
≤100/yr	341	99
≤250/yr	345	100

88

Appendix C

Sensitivity Analysis of ISC Air Model

C. Sensitivity Analysis of ISC Air Model

This appendix describes sensitivity analyses on depletion options, source shape and orientation, and receptor location and spacing.

C.1 Options With and Without Depletions

The Air Characteristic Study relies on the output of the air dispersion model ISCST3 to determine atmospheric concentrations of chemical constituents released by various waste management units. The ISCST3 model has several options for modeling deposition and depletion. Although the Air Characteristic Study is an inhalation-pathway-only analysis and does not require modeling deposition to estimate indirect pathway exposure, depletion is important. Depletion, or removal of chemical constituents from the plume due to deposition processes, can affect the estimate of air concentrations. Using ISCST3 with depletion, however, requires substantially more computer power and time to complete the extensive computer runs. Therefore, for the current study, depletion options are used only if the risk results are sensitive to the depletion option.

To determine ISCST3 model output sensitivity to depletion options for the Air Characteristic study, the magnitude of wet depletion of vapors and particulates and dry depletion of particulates were examined. ISCST3 cannot model dry depletion of vapors.

In this appendix, the setup of the sensitivity analysis is described, results are discussed, and recommendations are presented.

C.1.1 Setup of Sensitivity Analysis

Wastepiles and land application units (LAUs) were chosen to evaluate the effects of modeling air concentration of vapors and particulates with and without depletion. These two WMU types were selected to represent elevated and ground-level sources, respectively. Two areas were selected for each type of unit, one large and one small to capture differences in depletion due to source area. The two wastepile sizes were 20.25 m^2 and 1,300,056 m^2. The two land application unit sizes were 81 m^2 and 8,090,043 m^2. These sizes represent the smallest and largest size strata of each waste management unit type. Wastepiles were set to a source height of 5 meters and land application units were set to a source height of 0 meters. Receptors were placed at 25, 50, 75, 150, 500, and 1,000 meters from the edge of the source in concentric squares.

Meteorological data from Las Vegas, Nevada, and Miami, Florida, were selected for these analyses because these locations have the lowest and highest long-term average precipitation, respectively, of the 29 meteorological stations used in the Air Characteristic Study. Las Vegas averages 4.0 inches of precipitation per year and Miami averages 57.1 inches per year.

90

C.1.2 Analysis of Wet Depletion and Its Magnitude for Vapors and Particulates

The first sensitivity analysis was performed to evaluate the significance of wet depletion on calculated air concentrations of both vapors and particulates. The significance was determined by examining the magnitude of the difference between annual average air concentrations when wet depletion alone was selected vs. when both wet and dry depletion were selected. Both large and small wastepiles and land application units were run in this analysis, using the Hazardous Waste Identification Rule (HWIR 98) meteorological data and the 97363 version of ISCST3, which is the version of the model used for the May 1998 Air Characteristic Study. Because retention of the air dispersion model runs from the May 1998 study was desirable, the performance of this version of the model was important.

Because precipitation data in the SAMSON surface data set provided by the National Climatic Data Center are generally not complete, a specially processed meteorological data set was developed for the HWIR98 study based on precipitation data from the National Climatological Data Center (NCDC) Cooperative Station Summaries of the Day data set. Therefore, the new HWIR98 data set contains more complete precipitation data than were used in the May 1998 Air Characteristic Study and would yield a higher and more representative amount of wet depletion. The ISCST3 model was run for vapors and particulates with no depletion, wet depletion, and wet and dry depletion (particulates only) for both Miami and Las Vegas using those meteorological data.

Vapor Results. Vapors were evaluated for no depletion and wet depletion only because ISCST3 does not perform dry depletion calculations for vapor. In all cases, little difference in air concentration was observed when wet depletion was included versus excluded. Most differences were less than 1 percent, with lower air concentrations always resulting from the inclusion of wet depletion. Differences between air concentrations with and without wet depletion slightly exceeded 1 percent in a few cases when distances around 1 kilometer were examined. Only one receptor showed a difference of more than 2 percent. This receptor was for a small LAU in Miami at a distance of 1 kilometer. The difference was about 10 percent. The concentration values, however, are extremely small in magnitude due to the small size of the site. This difference is not considered significant.

Particulate Results. Large differences were observed between air concentrations with and without wet and dry depletion for particles for both LAUs and wastepiles. Maximum differences in air concentration ranged from 2 percent near the waste management unit to 57 percent at 1 kilometer from the waste management unit (see Tables C-1a and C-1b). The percent difference increases with increasing distance because the loss of material due to depletion is cumulative in nature. A comparison of air concentrations modeled with wet depletion only and those modeled with no depletion indicates that wet depletion plays a very small role in the annual average depletion of particles. Wet depletion alone yields less than 2 percent difference in concentration compared to no depletion.

Table C-1a. Maximum Percent Differences in Air Concentration with and without Wet Depletion Versus with and without Wet and Dry Depletion of Particles for Land Application Units

	Miami				Las Vegas			
	Large LAU		Small LAU		Large LAU		Small LAU	
Distance (m)	Wet Depletion	Wet/Dry Depletion	Wet Depletion	Wet/Dry Depletion	Wet Depletion	Wet/Dry Depletion	Wet Depletion	Wet/Dry Depletion
25	0.9	36.4	0.1	5.2	0.2	16.3	0.0	1.6
50	1.0	39.1	0.1	16.7	0.2	17.8	0.1	6.6
75	1.1	40.5	0.2	22.0	0.2	18.7	0.2	9.4
150	1.3	43.1	0.9	30.1	0.3	20.4	0.0	14.3
500	1.8	47.8	6.7	40.7	0.3	24.7	0.0	22.2
1000	2.2	51.0	20.0	50.0	0.4	29.1	0.0	33.3

Table C-1b. Maximum Percent Differences in Air Concentration with and without Wet Depletion Versus with and without Wet and Dry Depletion of Particles for Wastepiles

	Miami				Las Vegas			
	Large WP		Small WP		Large WP		Small WP	
Distance (m)	Wet Depletion	Wet/Dry Depletion	Wet Depletion	Wet/Dry Depletion	Wet Depletion	Wet/Dry Depletion	Wet Depletion	Wet/Dry Depletion
25	0.7	43.3	0.1	20.6	0.1	19.8	0.0	8.0
50	0.7	44.7	0.3	30.5	0.1	20.8	0.0	13.3
75	0.8	45.7	0.3	35.7	0.1	21.6	0.0	16.6
150	1.0	47.5	0.5	43.0	0.1	23.6	0.1	22.2
500	1.5	51.9	1.0	52.4	0.3	30.4	0.2	30.2
1000	2.0	57.0	1.6	56.3	0.4	34.8	0.3	34.0

92

C.1.3 Analysis of Different Versions of ISCST3 and Their Effects on Dry Depletion of Particles

A second analysis was performed to determine the difference between air concentration of particulates with dry depletion and without depletion for each of three different versions of ISCST3. The three versions were 96113 (1996), 97363 (1997), and 98356 (1998). The 1996 version of the model was used for the sensitivity analysis conducted as part of the May 1998 Air Characteristic Study. The 1997 version of the model was used for the production of the final results in May 1998 and is the model version used for the current study. The 1998 version, the latest release, was evaluated to determine if it should be adopted for the current study.

The algorithm for dry depletion differs in each of these versions of the model and the 1998 version contains a completely different area source integration routine from the other two versions. This analysis was performed using the large land application unit and the May 1998 Air Characteristic Study meteorological data. As shown in Section C.1.2., wet depletion is not significant for particles and thus is not included in this portion of the sensitivity analysis. Furthermore, there is no difference in the wet depletion algorithm among the three versions of ISCST3.

Results of Dry vs. No Depletion of Particles for Different Versions of ISCST3. As part of this analysis, differences in air concentration results produced by different versions of the ISCST3 model also were examined. When the three versions of ISCST3 were compared without depletion, no difference was found between the 1996 and 1997 versions of ISCST3. Small differences were found between these versions and the 1998 version and are not consistent in direction. These insignificant differences are due to changes in the area source integration technique[1] used by the model and modifications to some portions of the code.

There are, however, significant differences between the three versions of ISCST3 when dry depletion is included in the model run. The three versions were run for the largest land application unit strata (8,090,043 m^2). Large land application units were selected because larger differences between concentrations with and without depletion are generally expected at most receptor distances for a larger source. The dry depletion sensitivity analysis was conducted using all three versions of the ISCST3 model and the same meteorological data that were used in the May 1998 Air Characteristic Study.

The difference in air concentrations of particles with and without dry depletion for the 1996 version of ISCST3 was about 6 percent at 0 meters and increased to 25 percent at 1,000 meters for Las Vegas meteorological data. For Miami, these differences were 7 and 31 percent, respectively. The 1997 version showed differences of 12 and 34 percent at Las Vegas and 15 percent and 38 percent at Miami. The difference between the 1996 and 1997 models stems from the change in the deposition reference height from 20 times the roughness height to 1 meter.

[1]The integration technique arrives at a concentration value for a given receptor by performing a series of iterations, each representing the area source as consecutively increasing numbers of point emission sources, until there is little change in the result.

The difference in air concentrations of particles with and without depletion in the 1998 version of the model was larger. At Las Vegas, the difference was 22 percent at 0 meters and 46 percent at 1,000 meters. Miami showed 28 percent and 52 percent differences, respectively. These differences stem from changes to the algorithm used. These include a change in the integration technique and modifications that affected the vertical dispersion coefficient for area sources. Table C-1c shows the maximum differences with and without depletion for each version of ISCST3 at each distance from the source.

In addition to differences of the effect of depletion between the 1997 and 1998 versions of ISCST3, computer run time between these two versions of the model differed substantially. The 1998 version of the model required eight times the number of hours needed to complete a run than did the 1997 version of the model. The model developers were contacted for their recommendation about which version of the model to use. While they confirmed the increase in run time for the 1998 version, they offered no opinion as to which version is technically superior. Therefore, the 1997 version was chosen based on run time considerations and consistency with the previous Air Characteristic analysis.

C.1.4 Summary of Depletion Conclusions

Based on the results of these sensitivity analyses, the following decisions were made concerning the dispersion modeling:

- # Use meteorological data used in 1998 Air Characteristic Study since wet depletion will not be modeled for vapors or particulates.
- # Model vapors with no wet depletion because it is significant for vapors.
- # Model particulates with dry depletion only because wet depletion is not significant for particulates.
- # Use 1997 version of ISCST3 because it is technically valid, compatible with the May 1998 Air Characteristic Study, and the run time is substantially shorter than the run time for the 1998 version.

C.2 Source Shape and Orientation

A sensitivity analysis was conducted using the ISCST3 air model to determine what role source shape and orientation play in determining dispersion coefficients of air pollutants. A discussion of this analysis follows.

Three different sources were chosen for this analysis. The sources were a square (source No. 1), a rectangle oriented east to west (source No. 2), and a rectangle oriented north to south (source No. 3). All three sources had an area of 400 m^2 in order to ensure that equal emission rates were compared. The rectangles were selected to be exactly two times longer and half as wide as the square (see Figure C-2).

94

Table C-1c. Comparison of Maximum Percent Differences in Air Concentration of Particles with Versus Without Dry Depletion Between Three Versions of ISCST3

Distance (m)	Miami			Las Vegas		
	ISCST3 1996	ISCST3 1997	ISCST3 1998	ISCST3 1996	ISCST3 1997	ISCST3 1998
25	13	25	43	10	20	34
50	15	27	46	12	22	35
75	16	29	47	13	23	36
150	19	31	49	15	25	38
500	26	36	51	21	29	43
1000	31	39	53	25	34	47

Two meteorological stations at Little Rock, Arkansas, and Los Angeles, California, were selected for this modeling analysis in order to compare two different meteorological regimes. Little Rock was selected because of its evenly distributed wind directions and Los Angeles was selected because it has a predominantly southwest wind direction (see Figure C-3). Five years of meteorological data were used for this analysis.

Each area source was modeled with similar receptor grids to ensure consistency. Sixteen receptors were placed on the edge of each of the area sources and another 16 were placed 25 meters out from the edge. Each of these two receptor groups were modeled as a Cartesian receptor grid. Two receptor rings were also placed at 50 and 100 meters out from the center of the source. This polar receptor grid consisted of 16 receptors with a 22.5 degree interval between receptors. See Figures C-4a through C-4c for receptor locations.

The ISCST3 model was run using the meteorological data from Little Rock, Arkansas, and Los Angeles, California, and the results are shown in Tables C-2a and C-2b. The results indicated that the standard deviation of the differences in air concentrations is greatest between source No. 2 and source No. 3. This difference is due to the orientation of the source. This occurs for both the Cartesian receptor grid and the polar receptor grid at both meteorological locations. This shows that the model is sensitive to the orientation of the rectangular area source.

Standard deviations are significantly smaller when source No. 1 is compared to source Nos. 2 or 3. This shows that the differences in Unitized Air Concentration (UAC) between the square source and the two rectangular sources are less than the differences between the two rectangular sources. A square area source also contributes the least amount of impact of orientation. Since no information on source shape or orientation is available, a square source will minimize the errors caused by different source shapes and orientations.

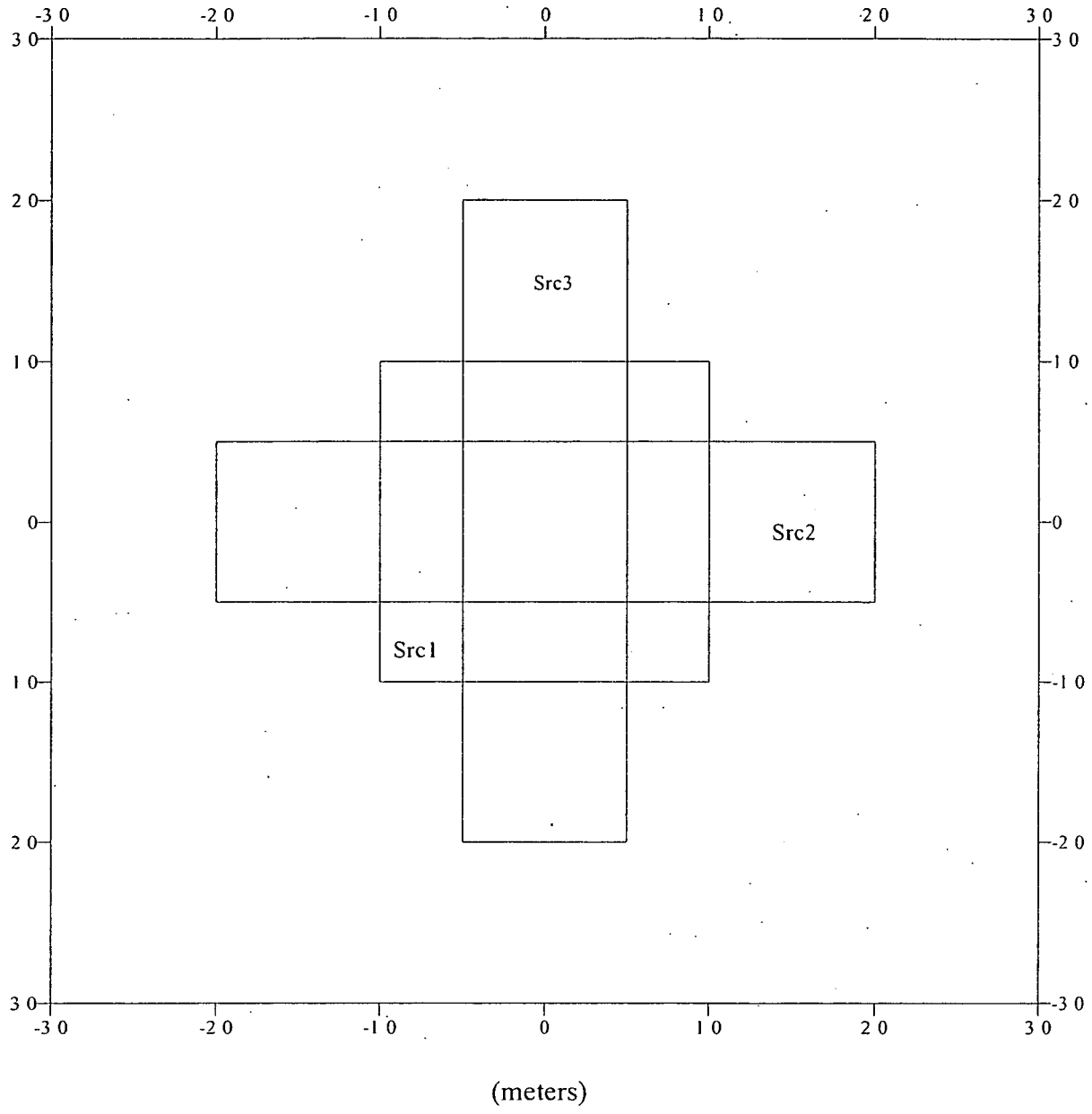
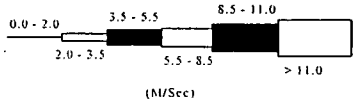
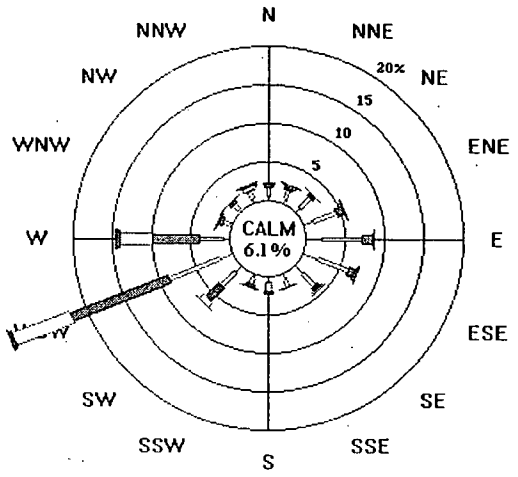


Figure C-2. Source Shapes and Orientations.

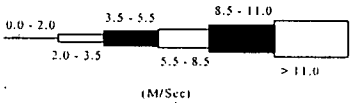
96

Los Angeles, California





Little Rock, Arkansas



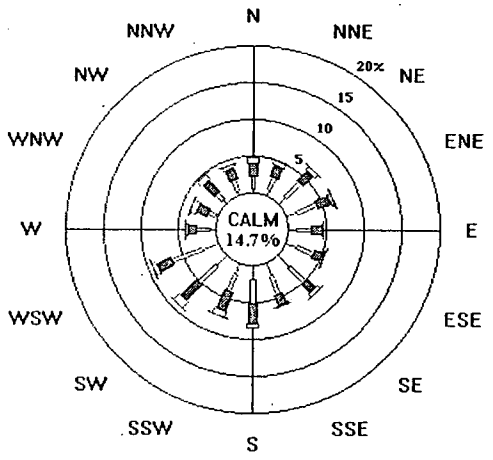


Figure C-3. Wind Roses

97

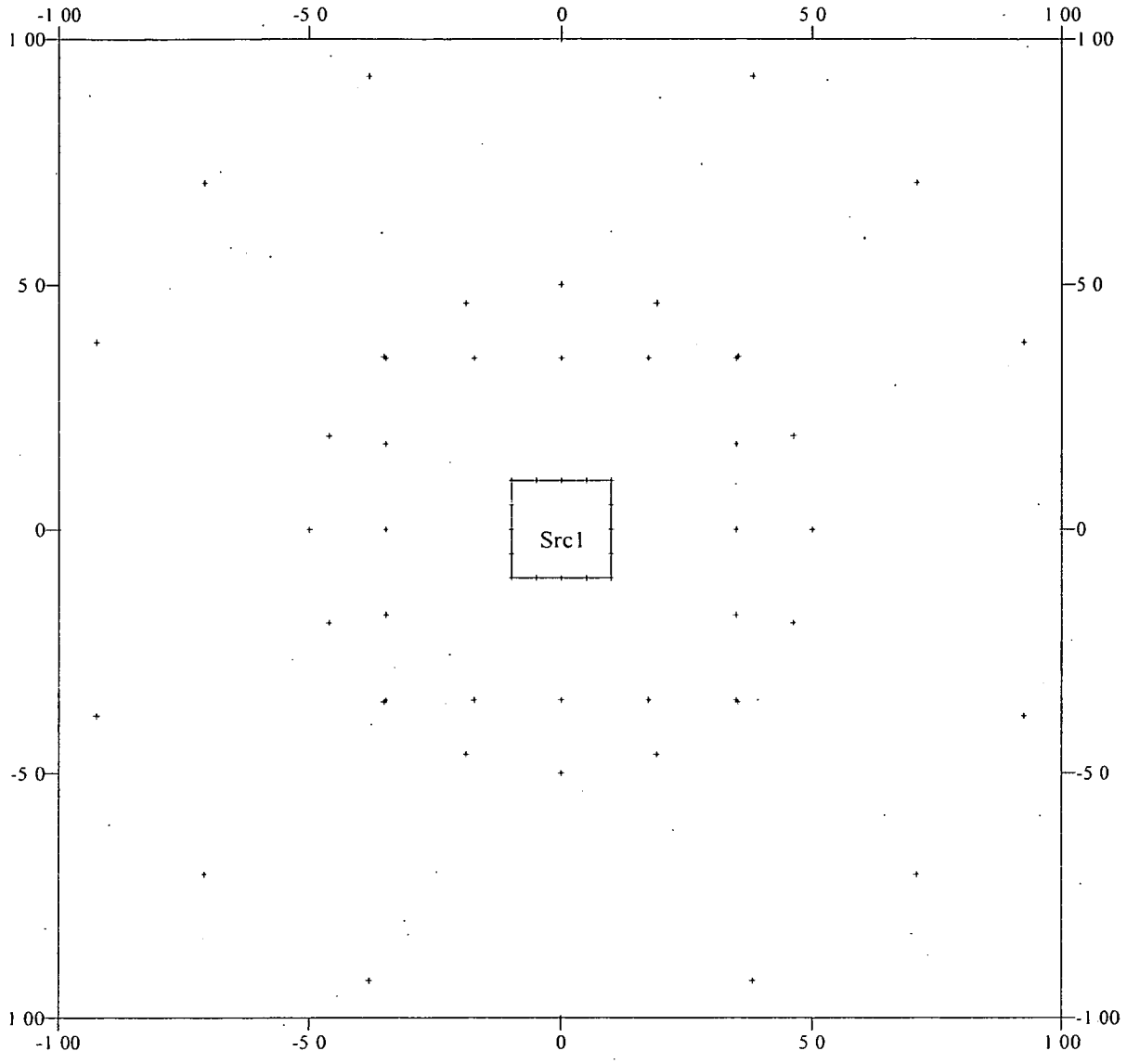
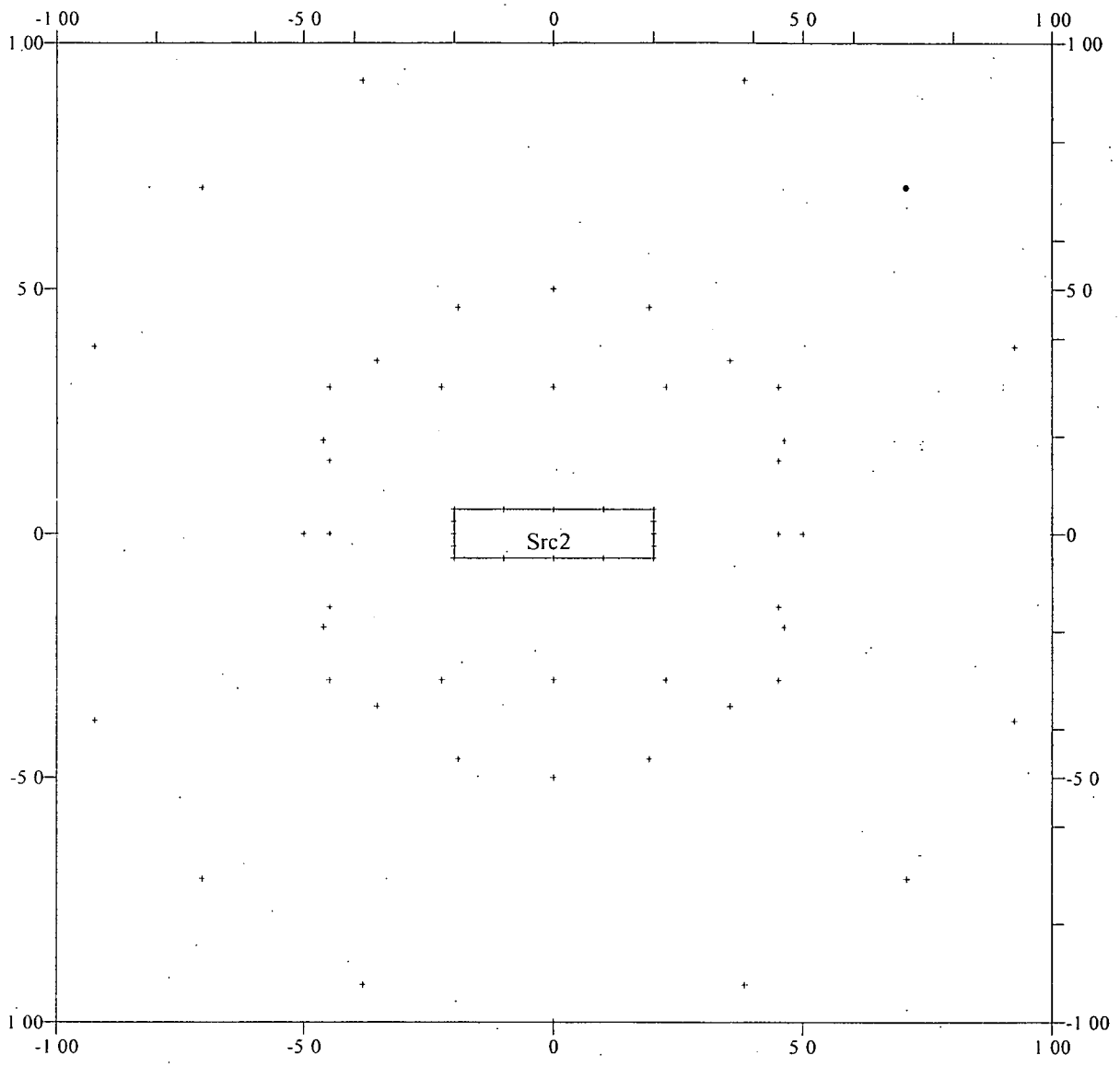


Figure C-4a. Receptor Locations (Source No. 1).

98



(meters)

Figure C-4b. Receptor Locations (Source No. 2).

99

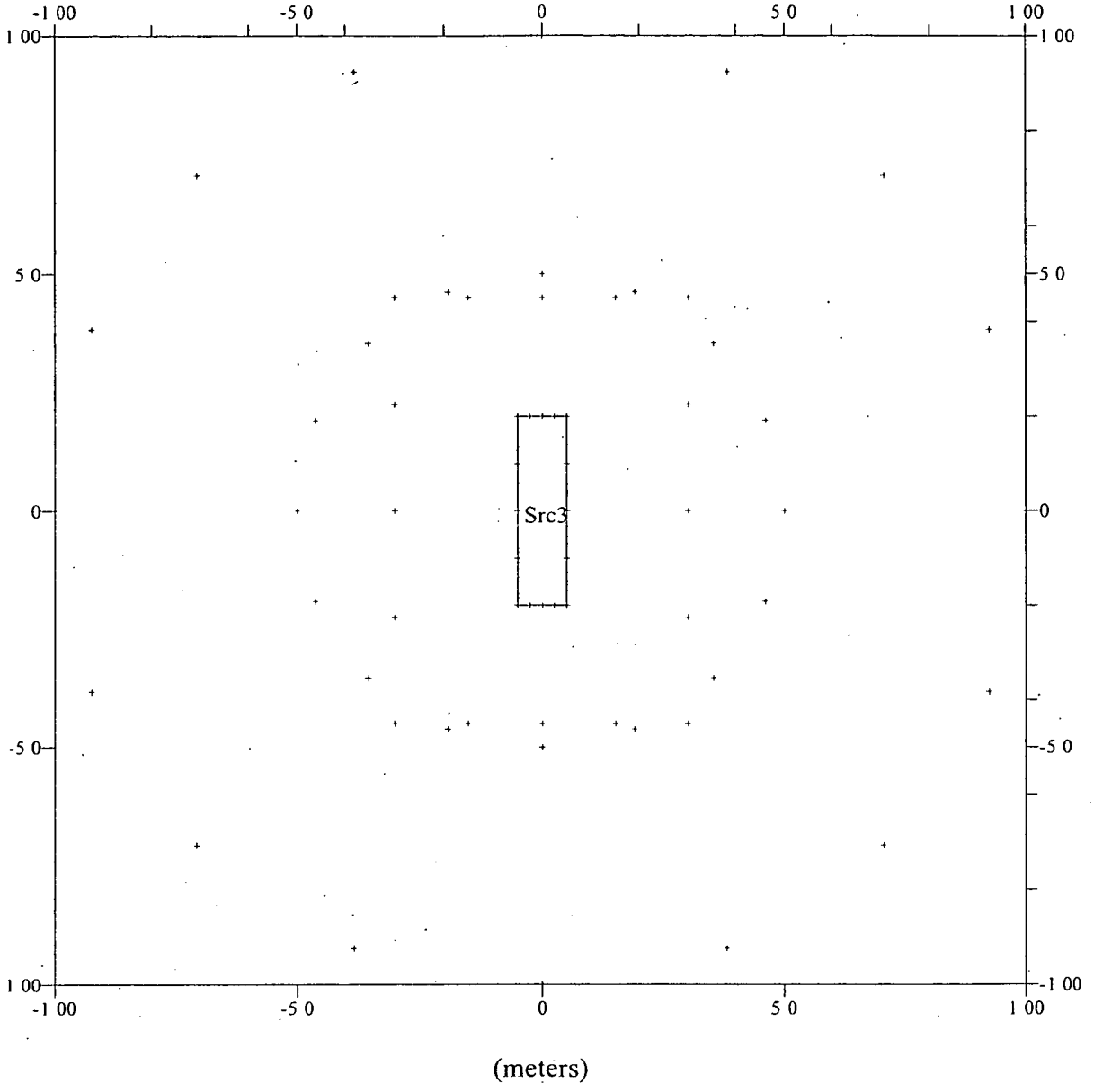


Figure C-4c. Receptor Locations (Source No. 3).

100

Table C-2a. Comparisons of Unitized Air Concentrations (µg/m³/µg/s-m²) for Different Source Shapes and Orientations (Little Rock, Arkansas)

Source No. 1 (20m x 20m) Polar Receptor Grid			Source No. 2 (40m x 10m)			Source No. 3 (10m x 40m)			Differences in UACs Sources No. 1 and No. 2		Differences in UACs Sources No. 1 and No. 3		Differences in UACs Sources No. 2 and No. 3	
X (m)	Y (m)	UAC	X (m)	Y (m)	UAC	X (m)	Y (m)	UAC	Diff. In UAC	% of Diff.	Diff. In UAC	% of Diff.	Diff. In UAC	% of Diff.
19	46	0.190	19	46	0.199	19	46	0.211	0.010	5%	0.021	11%	0.012	6%
38	92	0.050	38	92	0.051	38	92	0.051	0.001	1%	0.001	2%	0.000	1%
35	35	0.249	35	35	0.243	35	35	0.278	-0.007	-3%	0.028	11%	0.035	14%
71	71	0.067	71	71	0.067	71	71	0.069	-0.001	-1%	0.001	2%	0.002	3%
46	19	0.321	46	19	0.361	46	19	0.256	0.041	13%	-0.065	-20%	-0.105	-29%
92	38	0.095	92	38	0.098	92	38	0.088	0.003	3%	-0.007	-7%	-0.010	-10%
50	0	0.124	50	0	0.128	50	0	0.147	0.004	3%	0.023	19%	0.020	15%
100	0	0.030	100	0	0.030	100	0	0.033	0.000	-1%	0.003	9%	0.003	11%
46	-19	0.085	46	-19	0.096	46	-19	0.084	0.011	12%	-0.001	-1%	-0.011	-12%
92	-38	0.023	92	-38	0.024	92	-38	0.023	0.001	2%	-0.001	-2%	-0.001	-5%
35	-35	0.106	35	-35	0.109	35	-35	0.103	0.003	3%	-0.003	-3%	-0.006	-6%
71	-71	0.030	71	-71	0.030	71	-71	0.029	0.000	0%	0.000	-1%	-0.001	-2%
19	-46	0.117	19	-46	0.113	19	-46	0.128	-0.005	-4%	0.011	9%	0.016	14%
38	-92	0.033	38	-92	0.032	38	-92	0.034	-0.001	-4%	0.001	2%	0.002	7%
0	-50	0.122	0	-50	0.117	0	-50	0.143	-0.005	-4%	0.021	17%	0.026	22%
0	-100	0.035	0	-100	0.033	0	-100	0.037	-0.002	-5%	0.002	5%	0.004	11%
-19	-46	0.134	-19	-46	0.128	-19	-46	0.150	-0.006	-4%	0.016	12%	0.022	17%
-38	-92	0.038	-38	-92	0.036	-38	-92	0.038	-0.002	-4%	0.001	2%	0.002	6%
-35	-35	0.161	-35	-35	0.158	-35	-35	0.170	-0.003	-2%	0.009	6%	0.012	8%
-71	-71	0.043	-71	-71	0.043	-71	-71	0.045	0.000	1%	0.001	3%	0.001	3%
-46	-19	0.159	-46	-19	0.185	-46	-19	0.140	0.026	16%	-0.019	-12%	-0.045	-24%
-92	-38	0.044	-92	-38	0.046	-92	-38	0.043	0.002	4%	-0.002	-4%	-0.004	-8%
-50	0	0.103	-50	0	0.114	-50	0	0.107	0.011	11%	0.004	4%	-0.007	-6%
-100	0	0.027	-100	0	0.027	-100	0	0.027	0.000	2%	0.000	1%	0.000	0%
-46	19	0.126	-46	19	0.145	-46	19	0.118	0.019	15%	-0.008	-6%	-0.027	-18%
-92	38	0.035	-92	38	0.036	-92	38	0.034	0.001	4%	-0.001	-4%	-0.003	-7%
-35	35	0.152	-35	35	0.160	-35	35	0.153	0.008	5%	0.001	0%	-0.007	-5%
-71	71	0.041	-71	71	0.042	-71	71	0.041	0.001	3%	0.001	2%	-0.001	-2%
-19	46	0.173	-19	46	0.179	-19	46	0.187	0.007	4%	0.014	8%	0.008	4%
-38	92	0.047	-38	92	0.047	-38	92	0.048	0.000	0%	0.001	3%	0.001	3%
0	50	0.224	0	50	0.191	0	50	0.276	-0.032	-14%	0.052	23%	0.085	44%
0	100	0.068	0	100	0.061	0	100	0.074	-0.008	-11%	0.006	9%	0.014	22%
								Standard Deviation:	0.012	7%	0.018	9%	0.028	14%

(continued)

Table C-2a (continued)

Source No. 1 (20m x 20m)			Source No. 2 (40m x 10m)			Source No. 3 (10m x 40m)			Differences in UACs Sources No. 1 and No. 2		Differences in UACs Sources No. 1 and No. 3		Differences in UACs Sources No. 2 and No. 3	
Cartesion Receptor Grid														
X (m)	Y (m)	UAC	X (m)	Y (m)	UAC	X (m)	Y (m)	UAC	Diff. In UAC	% of Diff.	Diff. In UAC	% of Diff.	Diff. In UAC	% of Diff.
-10	-10	3.014	-20	-5	2.675	-5	-20	2.673	-0.339	-11%	-0.341	-11%	-0.002	0%
-5	-10	4.266	-10	-5	4.219	-2.5	-20	3.451	-0.047	-1%	-0.815	-19%	-0.769	-18%
0	-10	4.354	0	-5	4.307	0	-20	3.526	-0.047	-1%	-0.827	-19%	-0.781	-18%
5	-10	3.961	10	-5	4.069	2.5	-20	3.152	0.109	3%	-0.809	-20%	-0.918	-23%
10	-10	2.175	20	-5	1.899	5	-20	2.011	-0.276	-13%	-0.164	-8%	0.112	6%
10	-5	5.211	20	-2.5	3.875	5	-10	5.567	-1.337	-26%	0.355	7%	1.692	44%
10	0	5.968	20	0	4.704	5	0	5.913	-1.264	-21%	-0.055	-1%	1.209	26%
10	5	6.012	20	2.5	4.918	5	10	5.834	-1.094	-18%	-0.178	-3%	0.916	19%
10	10	4.946	20	5	4.468	5	20	4.344	-0.477	-10%	-0.602	-12%	-0.125	-3%
5	10	6.804	10	5	6.758	2.5	20	5.550	-0.047	-1%	-1.254	-18%	-1.208	-18%
0	10	6.846	0	5	6.830	0	20	5.604	-0.016	0%	-1.242	-18%	-1.226	-18%
-5	10	6.157	-10	5	6.353	-2.5	20	4.954	0.196	3%	-1.203	-20%	-1.399	-22%
-10	10	3.245	-20	5	2.793	-5	20	3.052	-0.451	-14%	-0.193	-6%	0.259	9%
-10	5	4.923	-20	2.5	3.801	-5	10	5.166	-1.121	-23%	0.244	5%	1.365	36%
-10	0	5.169	-20	0	4.032	-5	0	5.287	-1.137	-22%	0.118	2%	1.255	31%
-10	-5	4.809	-20	-2.5	3.727	-5	-10	4.991	-1.081	-22%	0.182	4%	1.264	34%
-35	-35	0.164	-45	-30	0.158	-30	-45	0.132	-0.006	-4%	-0.032	-19%	-0.026	-16%
-17.5	-35	0.219	-22.5	-30	0.247	-15	-45	0.167	0.027	12%	-0.052	-24%	-0.079	-32%
0	-35	0.243	0	-30	0.284	0	-45	0.179	0.041	17%	-0.063	-26%	-0.104	-37%
17.5	-35	0.186	22.5	-30	0.192	15	-45	0.147	0.006	3%	-0.039	-21%	-0.045	-23%
35	-35	0.108	45	-30	0.088	30	-45	0.100	-0.020	-19%	-0.008	-7%	0.012	14%
35	-17.5	0.141	45	-15	0.105	30	-22.5	0.160	-0.036	-25%	0.019	14%	0.055	52%
35	0	0.277	45	0	0.164	30	0	0.401	-0.113	-41%	0.124	45%	0.236	144%
35	17.5	0.503	45	15	0.396	30	22.5	0.466	-0.107	-21%	-0.037	-7%	0.070	18%
35	35	0.254	45	30	0.263	30	45	0.200	0.009	3%	-0.054	-21%	-0.063	-24%
17.5	35	0.315	22.5	30	0.373	15	45	0.234	0.058	18%	-0.081	-26%	-0.139	-37%
0	35	0.417	0	30	0.445	0	45	0.341	0.028	7%	-0.076	-18%	-0.104	-23%
-17.5	35	0.272	-22.5	30	0.286	-15	45	0.214	0.014	5%	-0.057	-21%	-0.071	-25%
-35	35	0.155	-45	30	0.131	-30	45	0.146	-0.024	-15%	-0.009	-6%	0.015	11%
-35	17.5	0.211	-45	15	0.155	-30	22.5	0.232	-0.056	-27%	0.022	10%	0.078	50%
-35	0	0.213	-45	0	0.145	-30	0	0.298	-0.068	-32%	0.084	40%	0.153	106%
-35	-17.5	0.265	-45	-15	0.193	-30	-22.5	0.264	-0.073	-27%	-0.002	-1%	0.071	37%
								Standard Deviation:	0.463	15%	0.435	17%	0.747	41%

102

Table C-2b. Comparisons of Unitized Air Concentrations ($\mu g/m^3/\mu g/s\text{-}m^2$) for Different Source Shapes and Orientations (Los Angeles, California)

Source No. 1 (20m x 20m)			Source No. 2 (40m x 10m)			Source No. 3 (10m x 40m)			Differences in UACs Sources No. 1 and No. 2		Differences in UACs Sources No. 1 and No. 3		Differences in UACs Sources No. 2 and No. 3	
Polar Receptor Grid														
X (m)	Y (m)	UAC	X (m)	Y (m)	UAC	X (m)	Y (m)	UAC	Diff. In UAC	% of Diff.	Diff. In UAC	% of Diff.	Diff. In UAC	% of Diff.
19	46	0.059	19	46	0.065	19	46	0.069	0.006	9%	0.010	17%	0.005	7%
38	92	0.016	38	92	0.016	38	92	0.016	0.000	-1%	0.000	3%	0.001	4%
35	35	0.188	35	35	0.168	35	35	0.284	-0.020	-11%	0.096	51%	0.116	69%
71	71	0.046	71	71	0.045	71	71	0.052	-0.001	-3%	0.006	13%	0.007	16%
46	19	0.582	46	19	0.607	46	19	0.461	0.025	4%	-0.121	-21%	-0.146	-24%
92	38	0.172	92	38	0.174	92	38	0.161	0.003	2%	-0.011	-6%	-0.014	-8%
50	0	0.278	50	0	0.293	50	0	0.293	0.014	5%	0.015	5%	0.001	0%
100	0	0.068	100	0	0.067	100	0	0.074	-0.001	-2%	0.005	8%	0.007	10%
46	-19	0.061	46	-19	0.062	46	-19	0.087	0.002	3%	0.026	43%	0.025	40%
92	-38	0.015	92	-38	0.015	92	-38	0.016	0.000	0%	0.002	10%	0.002	11%
35	-35	0.062	35	-35	0.068	35	-35	0.062	0.006	10%	0.000	0%	-0.006	-9%
71	-71	0.016	71	-71	0.017	71	-71	0.017	0.001	4%	0.001	3%	0.000	-1%
19	-46	0.080	19	-46	0.076	19	-46	0.087	-0.004	-4%	0.007	9%	0.011	14%
38	-92	0.023	38	-92	0.022	38	-92	0.024	-0.001	-5%	0.001	3%	0.002	8%
0	-50	0.086	0	-50	0.084	0	-50	0.096	-0.003	-3%	0.009	11%	0.012	15%
0	-100	0.023	0	-100	0.024	0	-100	0.024	0.000	1%	0.001	3%	0.000	2%
-19	-46	0.099	-19	-46	0.092	-19	-46	0.108	-0.006	-7%	0.009	9%	0.016	17%
-38	-92	0.028	-38	-92	0.027	-38	-92	0.028	-0.001	-2%	0.000	1%	0.001	3%
-35	-35	0.122	-35	-35	0.119	-35	-35	0.143	-0.003	-2%	0.021	18%	0.024	20%
-71	-71	0.033	-71	-71	0.032	-71	-71	0.034	0.000	-1%	0.001	4%	0.002	5%
-46	-19	0.218	-46	-19	0.223	-46	-19	0.226	0.005	2%	0.008	4%	0.003	2%
-92	-38	0.060	-92	-38	0.061	-92	-38	0.061	0.001	1%	0.001	1%	0.000	0%
-50	0	0.320	-50	0	0.378	-50	0	0.278	0.057	18%	-0.042	-13%	-0.099	-26%
-100	0	0.093	-100	0	0.098	-100	0	0.087	0.005	6%	-0.006	-6%	-0.011	-11%
-46	19	0.264	-46	19	0.273	-46	19	0.260	0.009	3%	-0.005	-2%	-0.013	-5%
-92	38	0.074	-92	38	0.075	-92	38	0.073	0.001	1%	-0.001	-2%	-0.002	-2%
-35	35	0.137	-35	35	0.123	-35	35	0.164	-0.014	-10%	0.027	20%	0.041	33%
-71	71	0.037	-71	71	0.035	-71	71	0.039	-0.002	-5%	0.002	4%	0.003	9%
-19	46	0.063	-19	46	0.066	-19	46	0.073	0.003	4%	0.010	15%	0.007	11%
-38	92	0.017	-38	92	0.017	-38	92	0.018	0.000	-2%	0.001	3%	0.001	5%
0	50	0.067	0	50	0.058	0	50	0.080	-0.008	-12%	0.014	21%	0.022	37%
0	100	0.020	0	100	0.018	0	100	0.021	-0.002	-9%	0.001	6%	0.003	15%
								Standard Deviation:	0.013	6%	0.030	14%	0.040	18%

(continued)

Table C-2b (continued)

Source No. 1 (20m x 20m)			Source No. 2 (40m x 10m)			Source No. 3 (10m x 40m)			Differences in UACs Sources No. 1 and No. 2		Differences in UACs Sources No. 1 and No. 3		Differences in UACs Sources No. 2 and No. 3	
Cartesion Receptor Grid														
X (m)	Y (m)	UAC	X (m)	Y (m)	UAC	X (m)	Y (m)	UAC	Diff. In UAC	% of Diff.	Diff. In UAC	% of Diff.	Diff. In UAC	% of Diff.
-10	-10	3.225	-20	-5	3.241	-5	-20	2.674	0.016	1%	-0.551	-17%	-0.567	-17%
-5	-10	4.025	-10	-5	4.333	-2.5	-20	3.119	0.308	8%	-0.906	-23%	-1.214	-28%
0	-10	3.952	0	-5	4.297	0	-20	3.050	0.345	9%	-0.902	-23%	-1.247	-29%
5	-10	3.431	10	-5	3.871	2.5	-20	2.564	0.440	13%	-0.867	-25%	-1.307	-34%
10	-10	1.683	20	-5	1.592	5	-20	1.511	-0.091	-5%	-0.172	-10%	-0.081	-5%
10	-5	5.931	20	-2.5	4.787	5	-10	5.570	-1.143	-19%	-0.360	-6%	0.783	16%
10	0	6.636	20	0	5.882	5	0	5.644	-0.754	-11%	-0.992	-15%	-0.238	-4%
10	5	6.640	20	2.5	6.294	5	10	5.524	-0.346	-5%	-1.116	-17%	-0.770	-12%
10	10	5.600	20	5	5.866	5	20	4.325	0.266	5%	-1.275	-23%	-1.541	-26%
5	10	6.893	10	5	8.126	2.5	20	4.939	1.232	18%	-1.955	-28%	-3.187	-39%
0	10	6.860	0	5	8.285	0	20	4.913	1.424	21%	-1.947	-28%	-3.371	-41%
-5	10	6.031	-10	5	7.442	-2.5	20	4.156	1.411	23%	-1.875	-31%	-3.286	-44%
-10	10	3.393	-20	5	3.497	-5	20	2.702	0.103	3%	-0.691	-20%	-0.794	-23%
-10	5	5.649	-20	2.5	5.102	-5	10	5.015	-0.547	-10%	-0.634	-11%	-0.088	-2%
-10	0	5.944	-20	0	5.373	-5	0	5.167	-0.572	-10%	-0.777	-13%	-0.205	-4%
-10	-5	5.663	-20	-2.5	5.028	-5	-10	5.104	-0.635	-11%	-0.559	-10%	0.076	2%
-35	-35	0.124	-45	-30	0.139	-30	-45	0.095	0.014	11%	-0.029	-23%	-0.043	-31%
-17.5	-35	0.158	-22.5	-30	0.183	-15	-45	0.123	0.025	16%	-0.035	-22%	-0.060	-33%
0	-35	0.172	0	-30	0.199	0	-45	0.121	0.028	16%	-0.050	-29%	-0.078	-39%
17.5	-35	0.123	22.5	-30	0.124	15	-45	0.100	0.001	0%	-0.024	-19%	-0.024	-20%
35	-35	0.064	45	-30	0.053	30	-45	0.063	-0.011	-17%	-0.001	-2%	0.010	19%
35	-17.5	0.095	45	-15	0.076	30	-22.5	0.119	-0.019	-20%	0.024	25%	0.043	57%
35	0	0.592	45	0	0.377	30	0	0.696	-0.215	-36%	0.104	18%	0.319	85%
35	17.5	0.829	45	15	0.739	30	22.5	0.683	-0.090	-11%	-0.146	-18%	-0.055	-7%
35	35	0.192	45	30	0.304	30	45	0.101	0.112	58%	-0.091	-47%	-0.203	-67%
17.5	35	0.109	22.5	30	0.195	15	45	0.072	0.086	78%	-0.037	-34%	-0.122	-63%
0	35	0.125	0	30	0.144	0	45	0.100	0.019	15%	-0.025	-20%	-0.044	-31%
-17.5	35	0.113	-22.5	30	0.160	-15	45	0.077	0.047	42%	-0.035	-31%	-0.082	-52%
-35	35	0.139	-45	30	0.166	-30	45	0.089	0.026	19%	-0.050	-36%	-0.077	-46%
-35	17.5	0.387	-45	15	0.335	-30	22.5	0.370	-0.053	-14%	-0.017	-4%	0.036	11%
-35	0	0.603	-45	0	0.472	-30	0	0.603	-0.131	-22%	0.000	0%	0.131	28%
-35	-17.5	0.318	-45	-15	0.275	-30	-22.5	0.316	-0.043	-13%	-0.002	-1%	0.041	15%
								Standard Deviation:	0.542	24%	0.614	15%	1.026	33%

104

C.3 Receptor Locations and Spacings

A sensitivity analysis was conducted using the ISCST3 model to determine what receptor locations and spacings should be used in the risk analysis for five types of waste management units (WMUs). A discussion of the analysis follows.

Because it takes a substantial amount of time for the ISCST3 model to execute, it was necessary to choose a limited number of receptors to be used in the dispersion modeling analysis. The larger the number of receptor points, the longer the run time. However, modeling fewer receptors may result in the omission of the maximum point for assessing exposure impacts. Therefore, a sensitivity analysis was conducted to determine the number of receptors needed for the model run and to locate ideal receptor placements.

A wind rose was plotted for each of the 29 meteorological stations to be used in the risk analysis for a 5-year time period in order to choose two meteorological stations for this sensitivity analysis. Little Rock, Arkansas, and Los Angeles, California, meteorological stations were selected for the sensitivity analysis. The wind roses show that Little Rock has very evenly distributed wind directions, and Los Angeles has a predominant southwest to west wind (Figure C-3). Little Rock and Los Angeles were chosen to determine if a higher density of receptors should be placed downwind of a site near Los Angeles, as compared to a site near Little Rock. Similarly, the 5th, 50th, and 95th percentile of sizes of LAUs were used in the sensitivity analysis to determine whether sizes of units can affect receptor locations and spacings. The areas of the 5th, 50th, and 95th percentile of sizes of LAUs are 1,200 m^2, 100,000 m^2, and 1,700,000 m^2, respectively.

The dispersion modeling was conducted using two sets of receptor grids. The first set of receptor points (Cartesian receptor grid) was placed around the modeled source with distances of 25, 50, 75, and 150 meters from the edge of the unit. Square-shaped ground-level area sources were used in the modeling. Therefore, these receptors are located on five squares surrounding the source. The second set of receptor points (polar receptor grid) was placed outside of the first set of receptors to 10 kilometers from the center of the source. Since the ISCST3 model's area source algorithm does not consider elevated terrain, receptor elevations were not input in the modeling.

In this sensitivity analysis, both downwind and lateral receptor spacings were investigated for three unit sizes using 5 years of meteorological data from Little Rock and Los Angeles. For the first set of receptor points (i.e., Cartesian receptor grid), five downwind distances of 25, 50, 75, and 150 meters from the edge of the source were used. For lateral receptor spacing, choices of 64, 32, and 16 equally spaced receptor points for each square were used in the modeling to determine the number of receptors needed to catch the maximum impacts. (See Figures C-5a through C-5c for Cartesian receptor locations and spacings [50th percentile]). For the second set of receptor points (i.e., polar receptor grid), about 20 downwind distances (i.e., receptor rings) were used. Receptor lateral intervals of 22.5° and 10° were used to determine whether 22.5° spacing can catch the maximum impacts. With a 22.5° interval, there are 16 receptors on each ring. There are 36 receptors on each ring for the 10° interval. See Figures C-6a and C-6b for polar receptor locations (5th percentile).

The results (Figures C-7a through C-7f) show that the maximum downwind concentrations decrease sharply from the edge of the area source to 150 meters from the source. The maximum concentrations decrease more sharply for a smaller area source than for a larger one. This means that more close-to-source receptors are generally needed for a small area source than for a large one.

The results also show that the maximum impacts are generally higher for a dense receptor grid (i.e., 64 or 32 receptors on each square) than for a scattered receptor grid (i.e., 16 receptors on each square). However, the differences of the maximum receptor impacts are not significant between a dense and a scattered receptor grid (Figures C-7a through C-7f). It should be noted that the above conclusions apply to both Little Rock and Los Angeles. This means that the distribution of wind directions does not play an important role in determining receptor lateral spacings.

Figures C-8a through C-8f compare the maximum concentrations at each ring for 22.5° and 10° intervals. The results show that the differences of the maximum concentrations are greater for close-to-source receptors than for further out receptors, and the differences are greater for larger area sources than for smaller area sources. The differences of the maximum concentrations for 22.5° and 10° intervals are generally small, and the concentrations tend to be the same at 10 kilometers. The conclusions were drawn from both Little Rock and Los Angeles meteorological data.

106

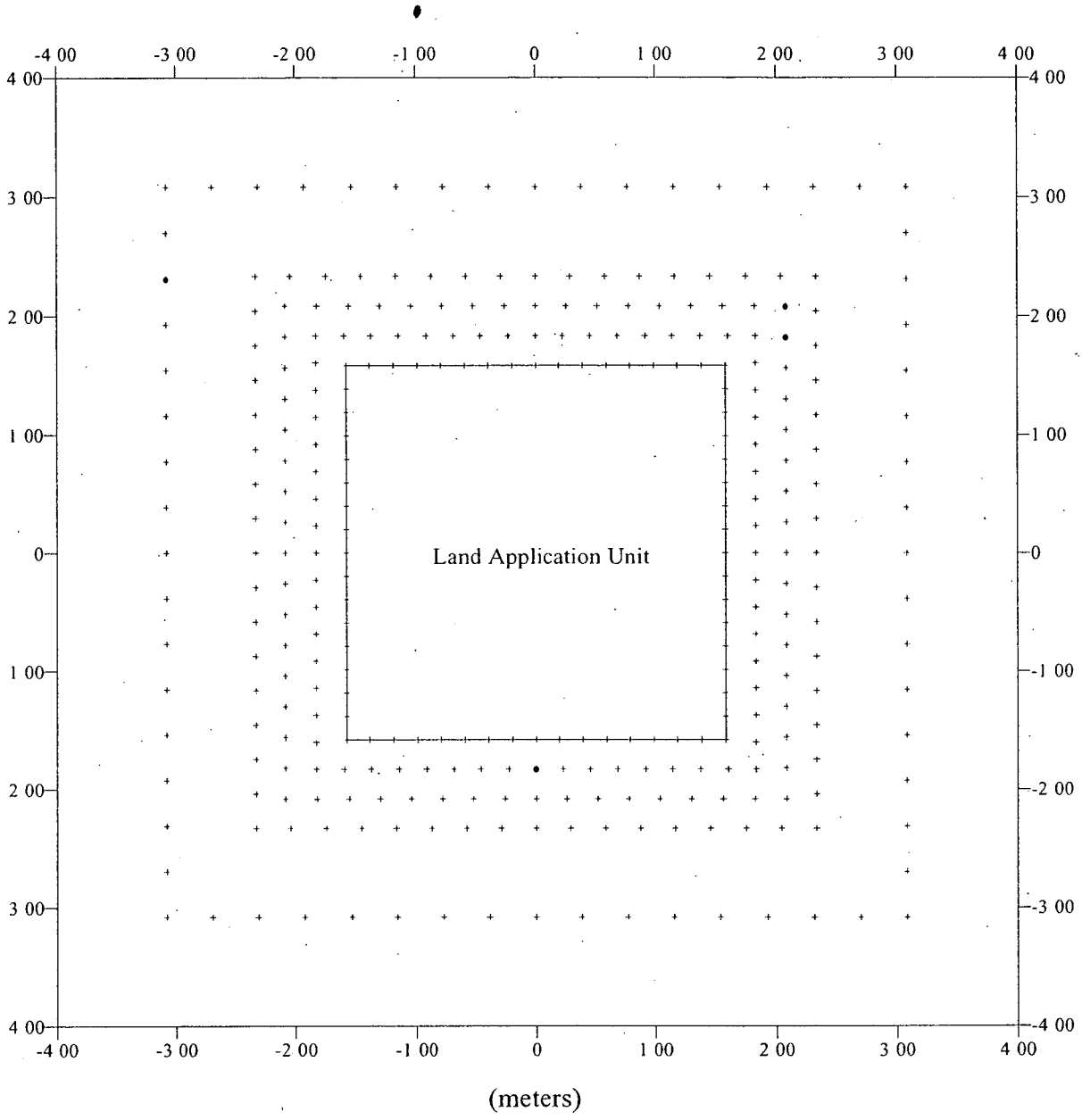


Figure C-5a. Cartesian Receptor Grid (64 receptors each square).

107

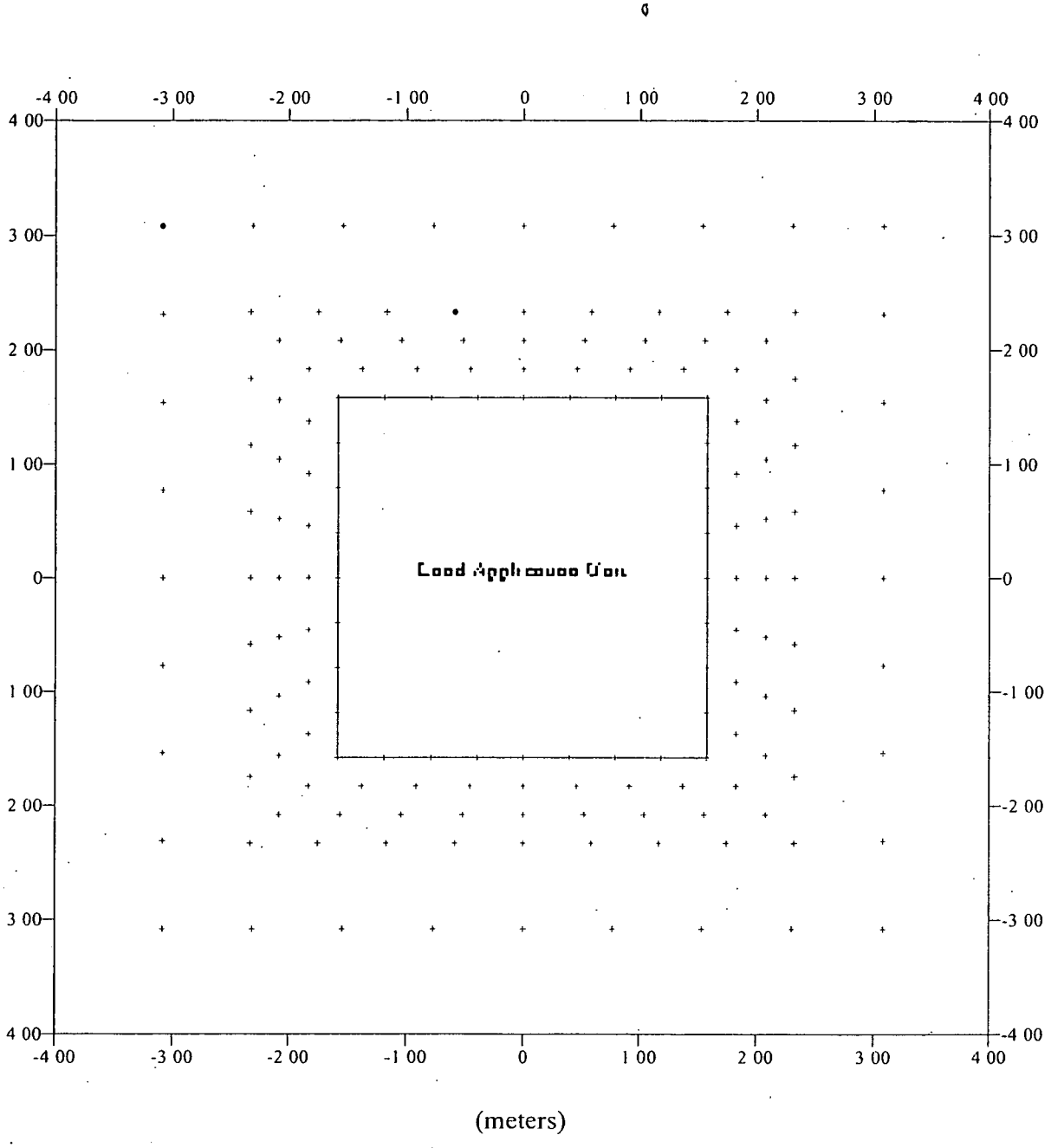


Figure C-5b. Cartesian Receptor Grid (32 receptors each square).

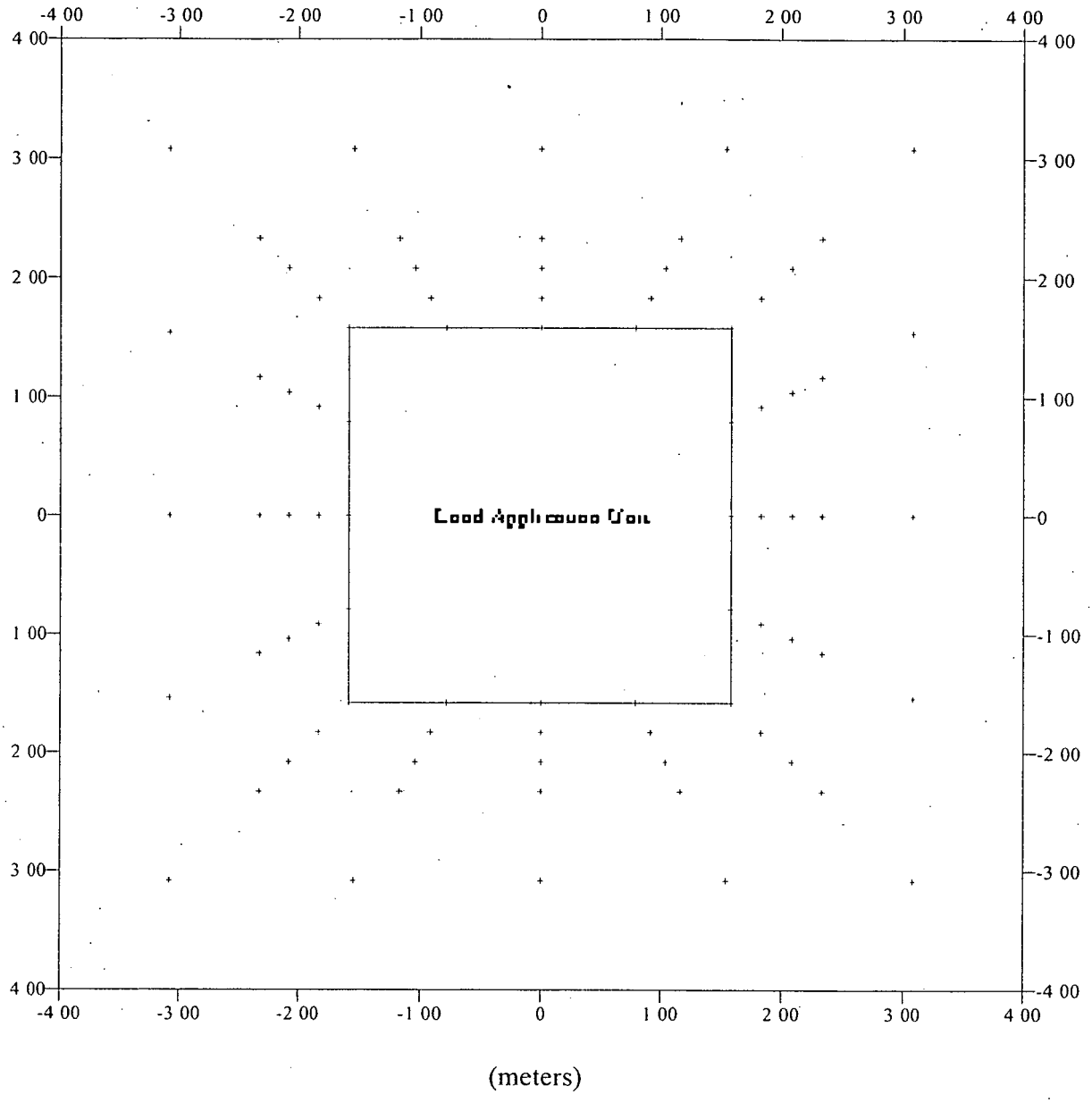


Figure C-5c. Cartesian Receptor Grid (16 receptors each square).

109

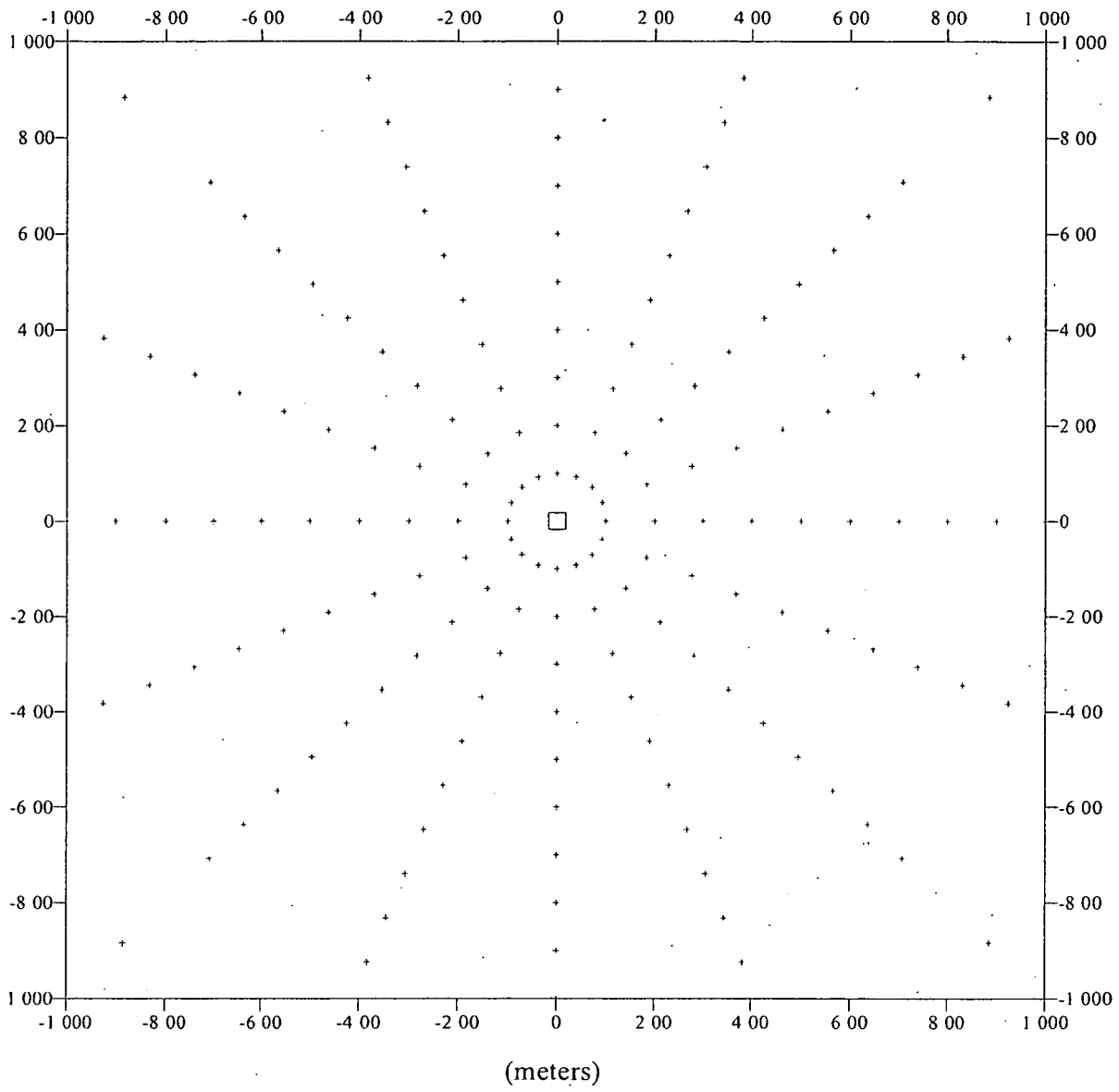


Figure C-6a. Polar Receptor Grid (22.5 degree).

110

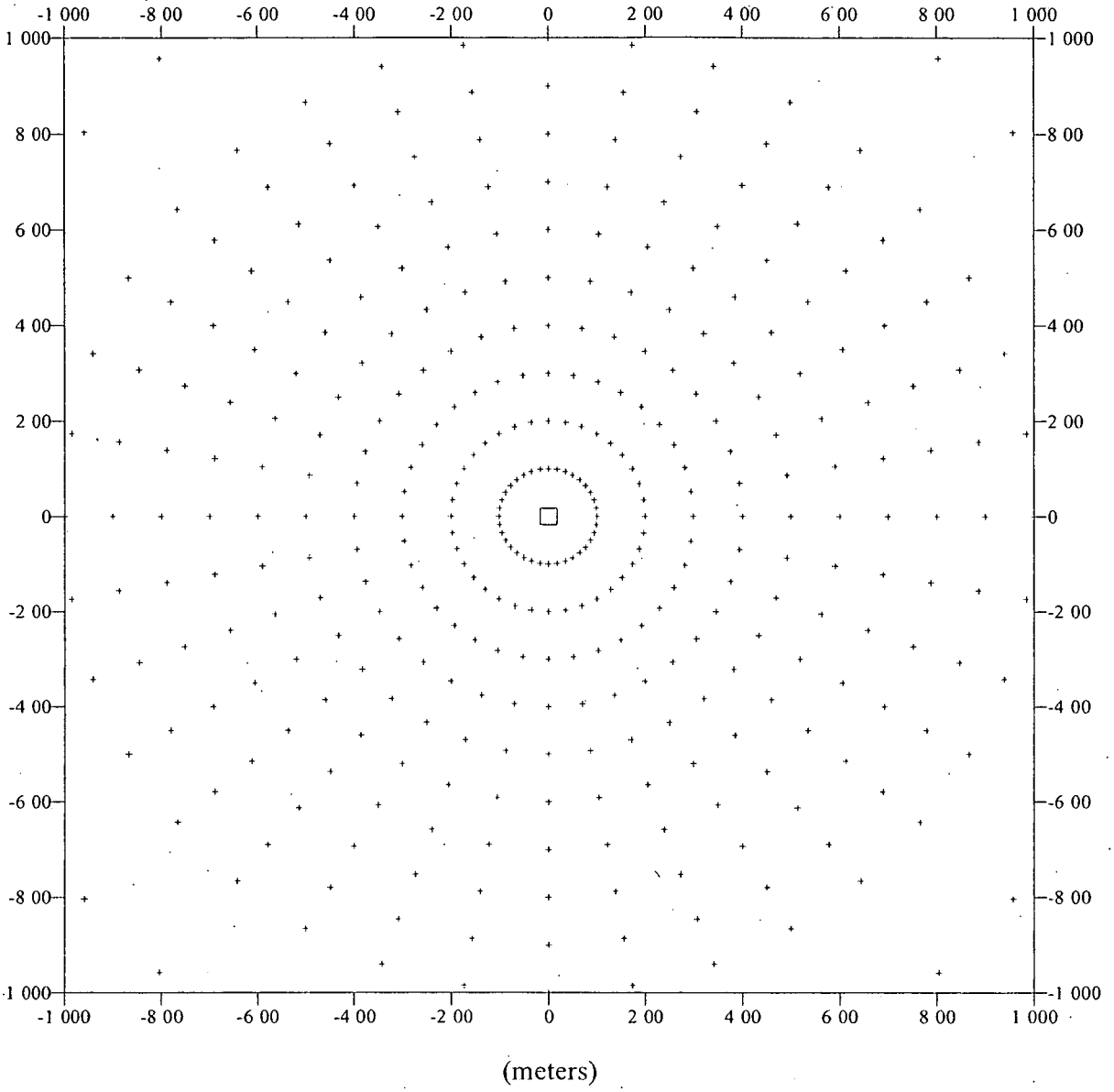


Figure D-6b. Polar Receptor Grid (10 degree).

111

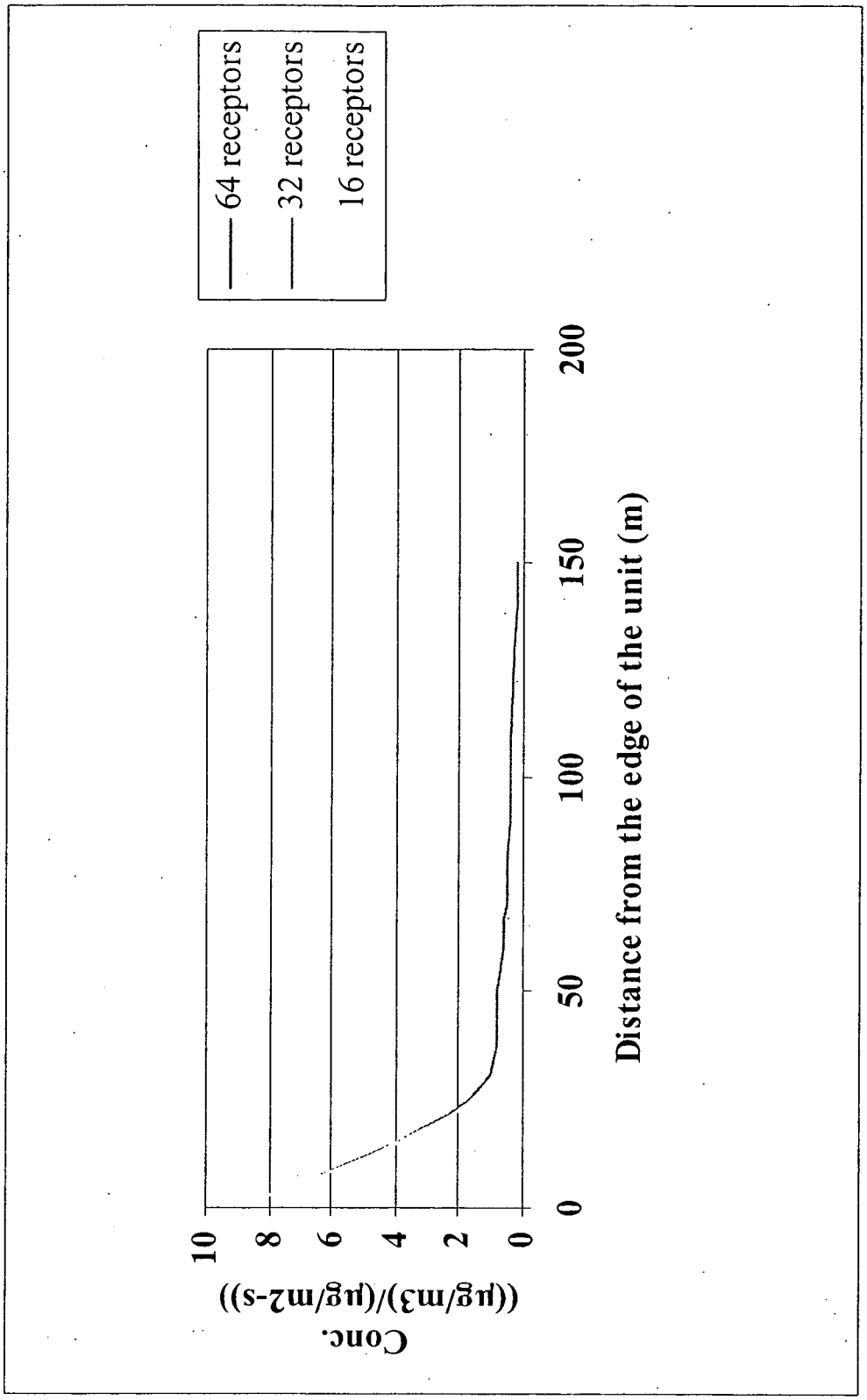


Figure C-7a. Maximum Concentrations (5th Percentile, LAU, Los Angeles, CA)

112

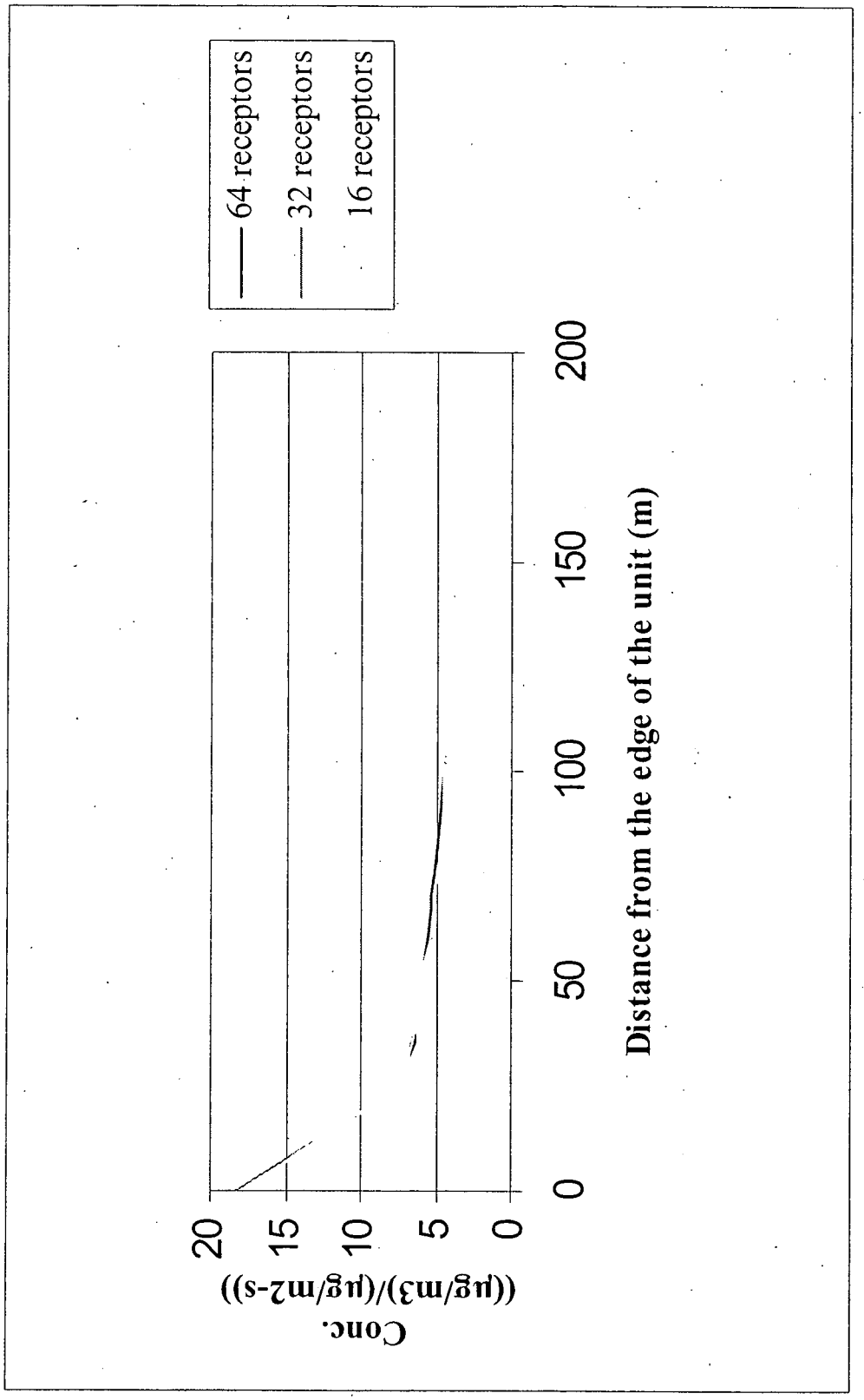


Figure C-7b. Maximum Concentrations (50th Percentile, LAU, Los Angeles, CA)

113

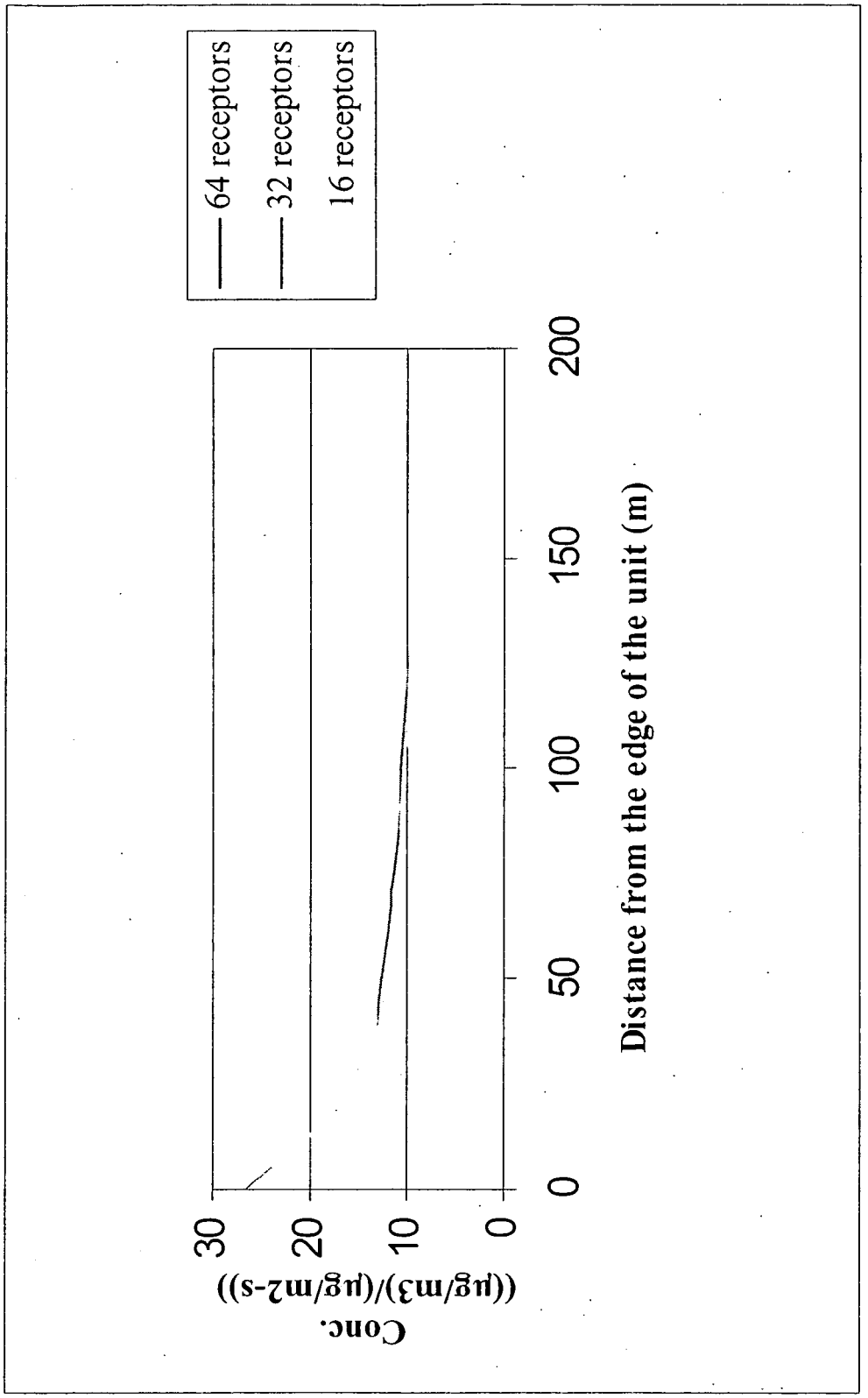


Figure C-7c. Maximum Concentrations (95th Percentile, LAU, Los Angeles, CA)

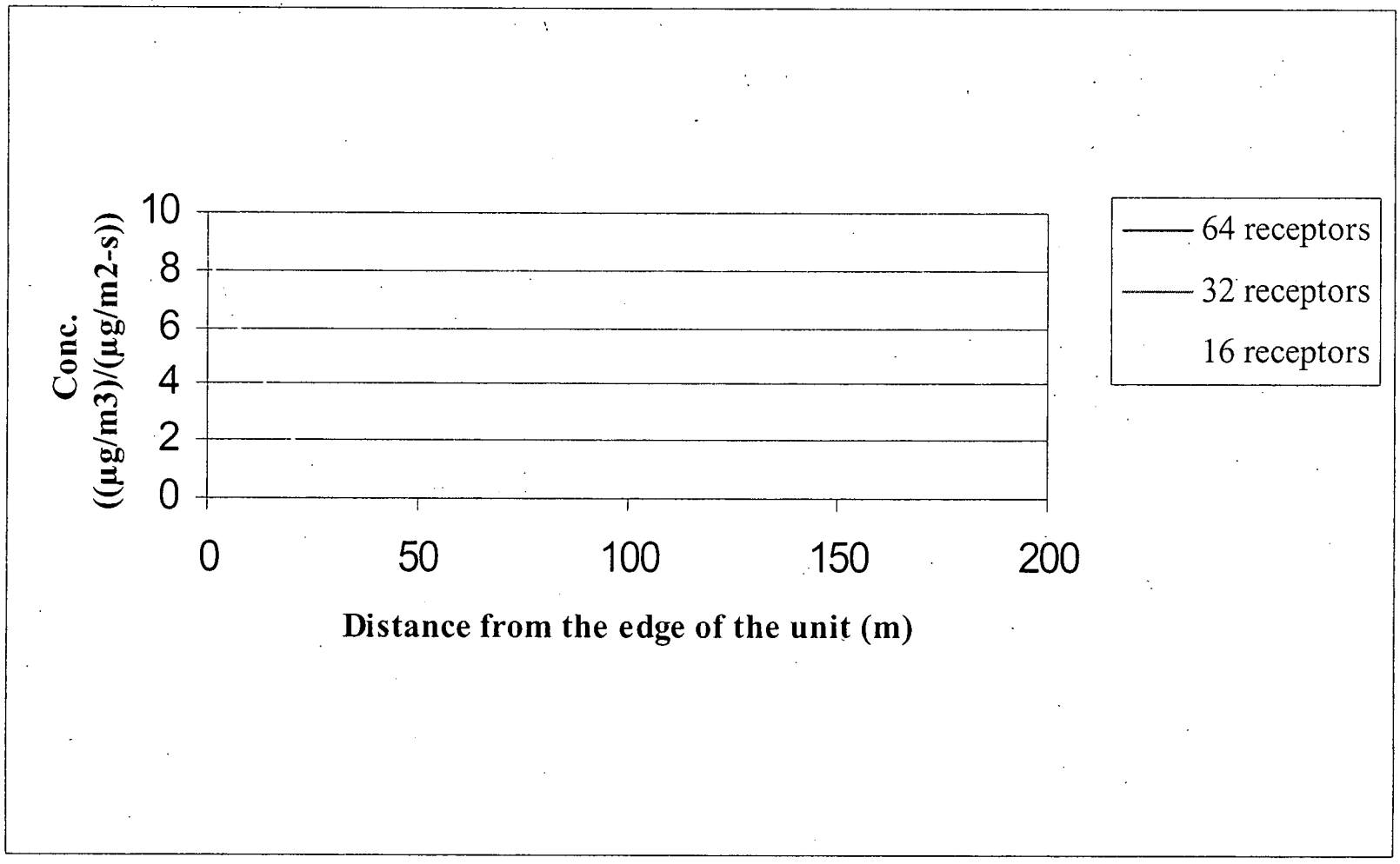


Figure C-7d. Maximum Concentrations (5th Percentile, LAU, Little Rock, AR)

115

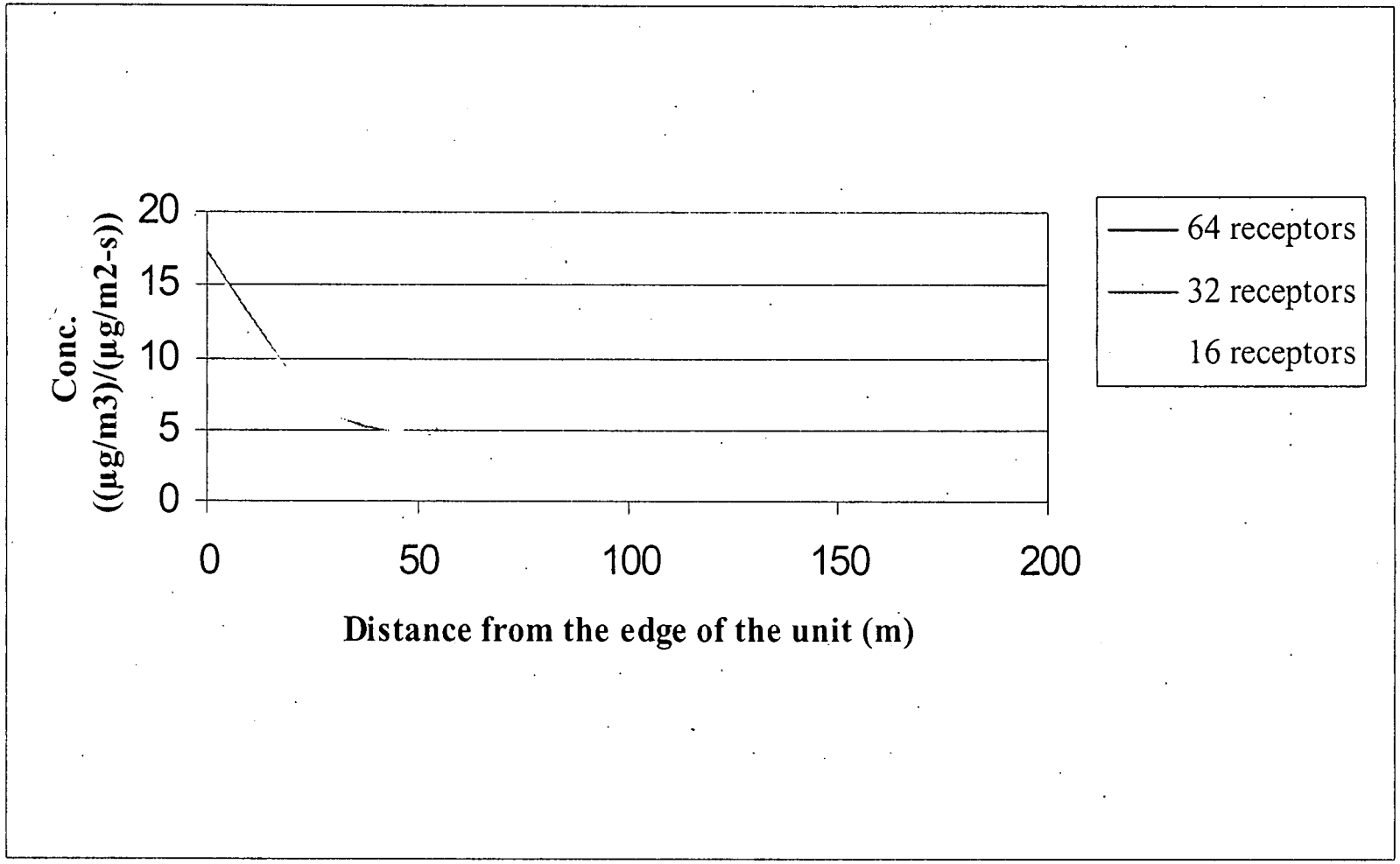


Figure C-7e. Maximum Concentrations (50th Percentile, LAU, Little Rock, AR)

116

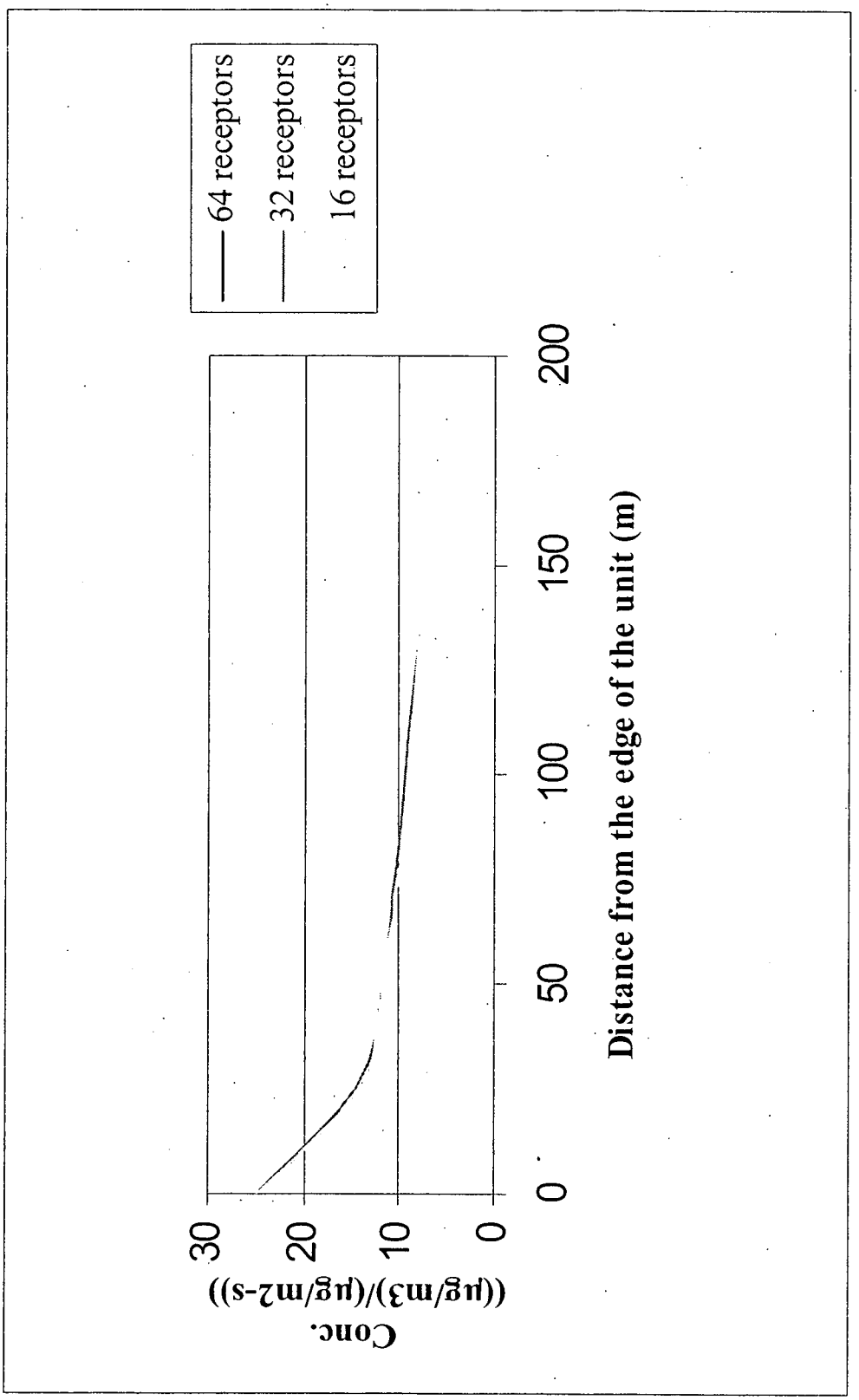


Figure C-7f. Maximum Concentrations (95th Percentile, LAU, Little Rock, AR)

117

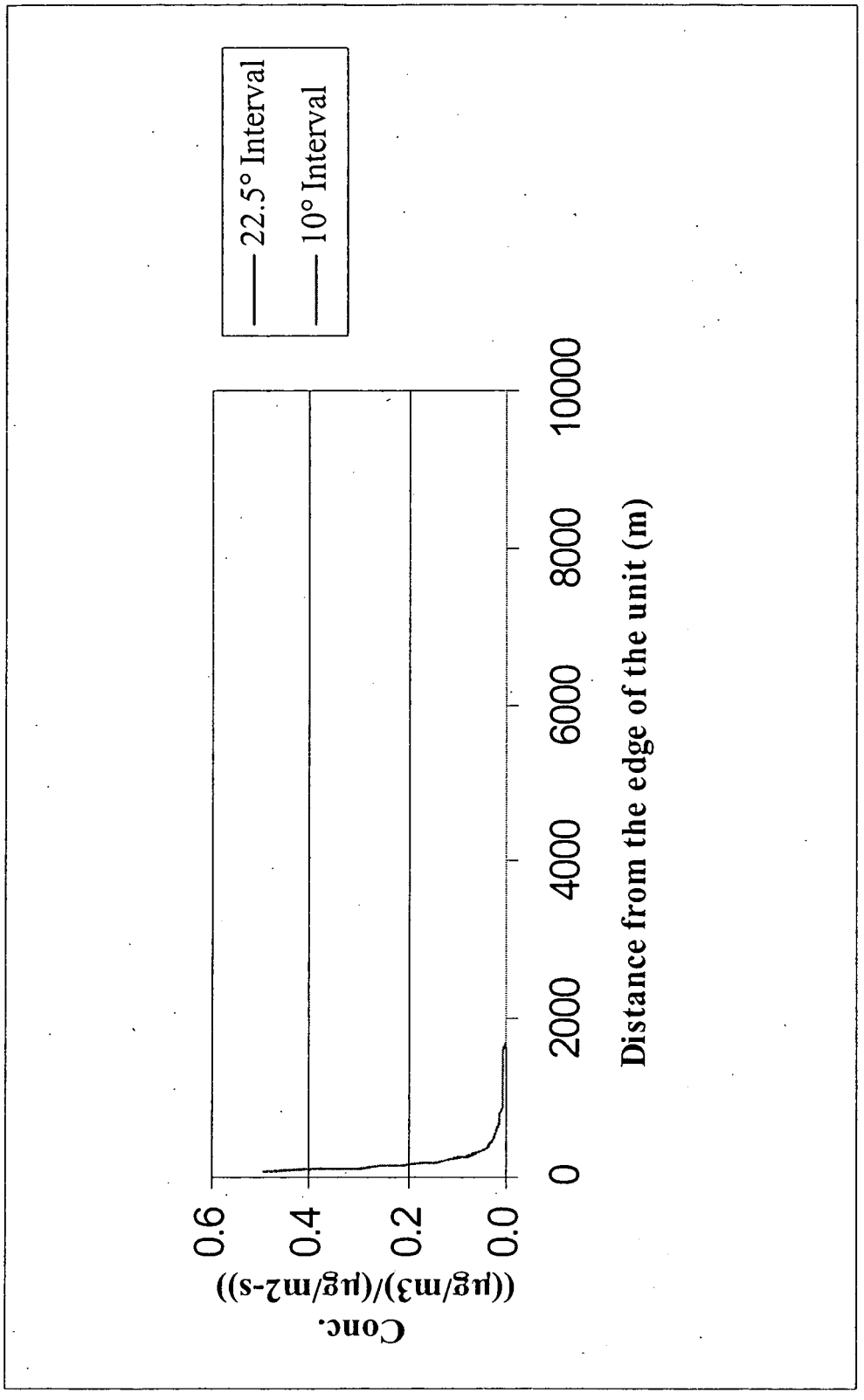


Figure C-8a. Maximum Concentrations (5th Percentile, LAU, Lose Angeles, CA)

118

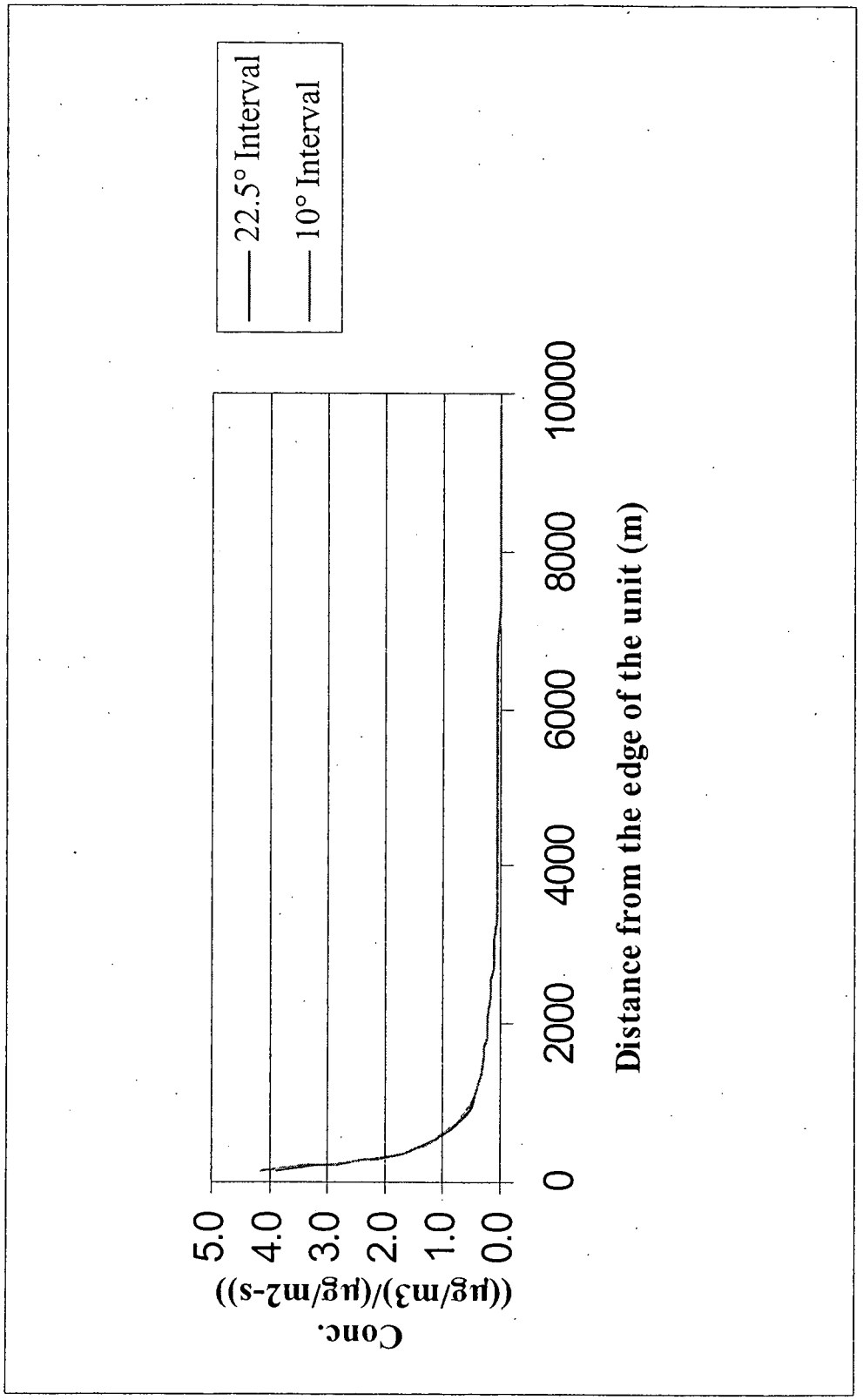


Figure C-8b. Maximum Concentrations (50th Percentile, LAU, Los Angeles, CA)

119

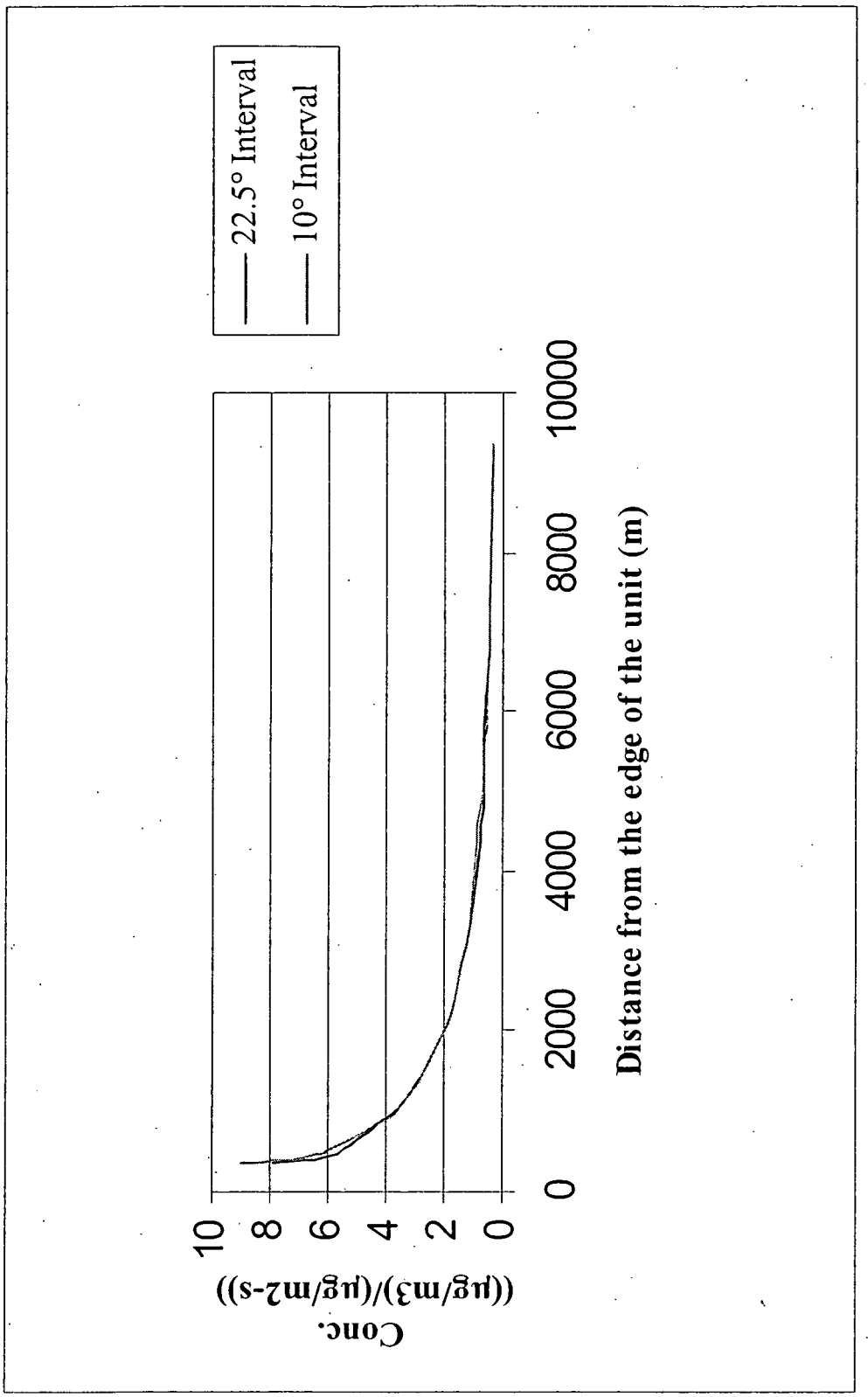


Figure C-8c. Maximum Concentrations (95th Percentile, LAU, Los Angeles, CA)

120

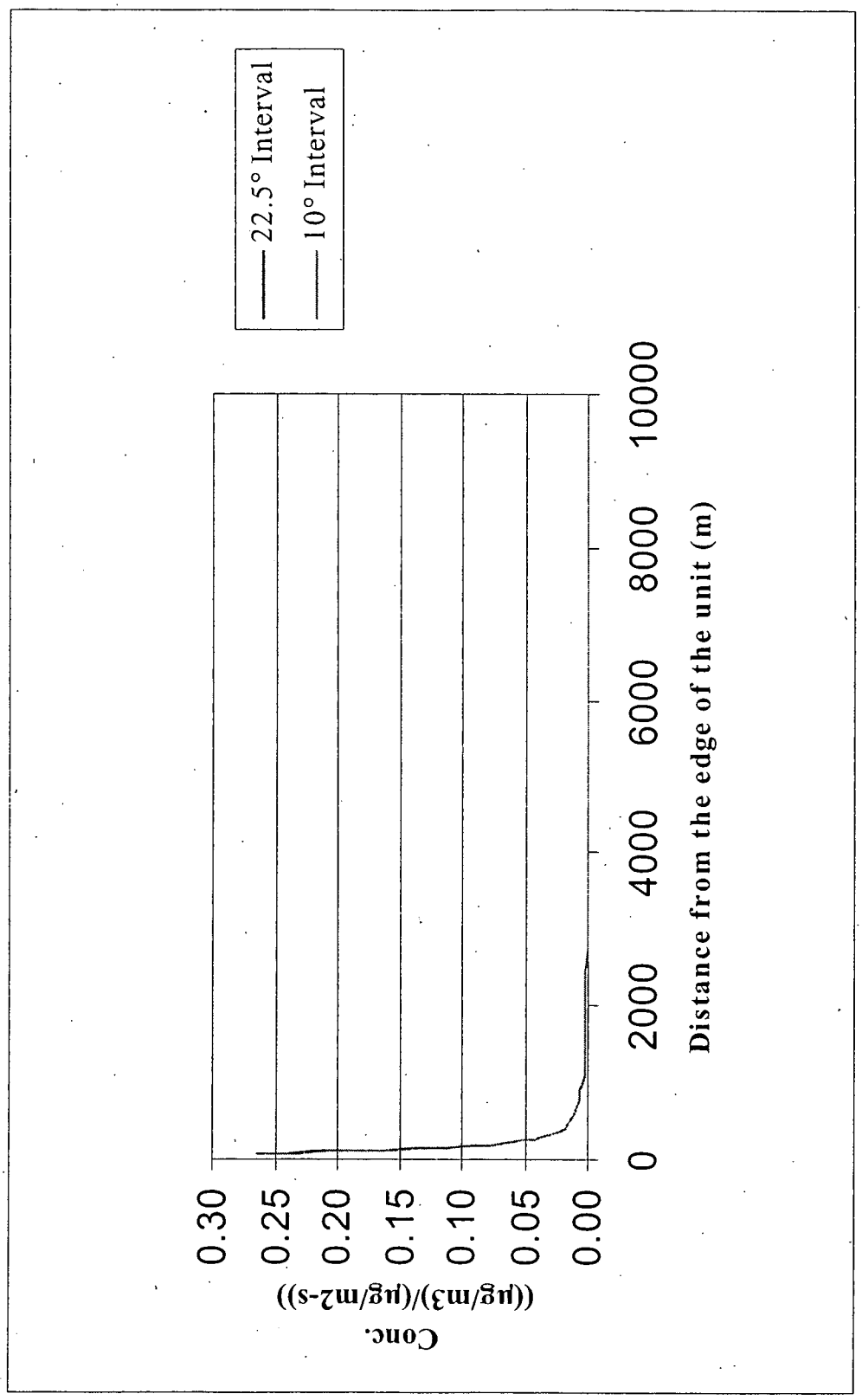


Figure C-8d. Maximum Concentrations (5th Percentile, LAU, Little Rock, AR)

121

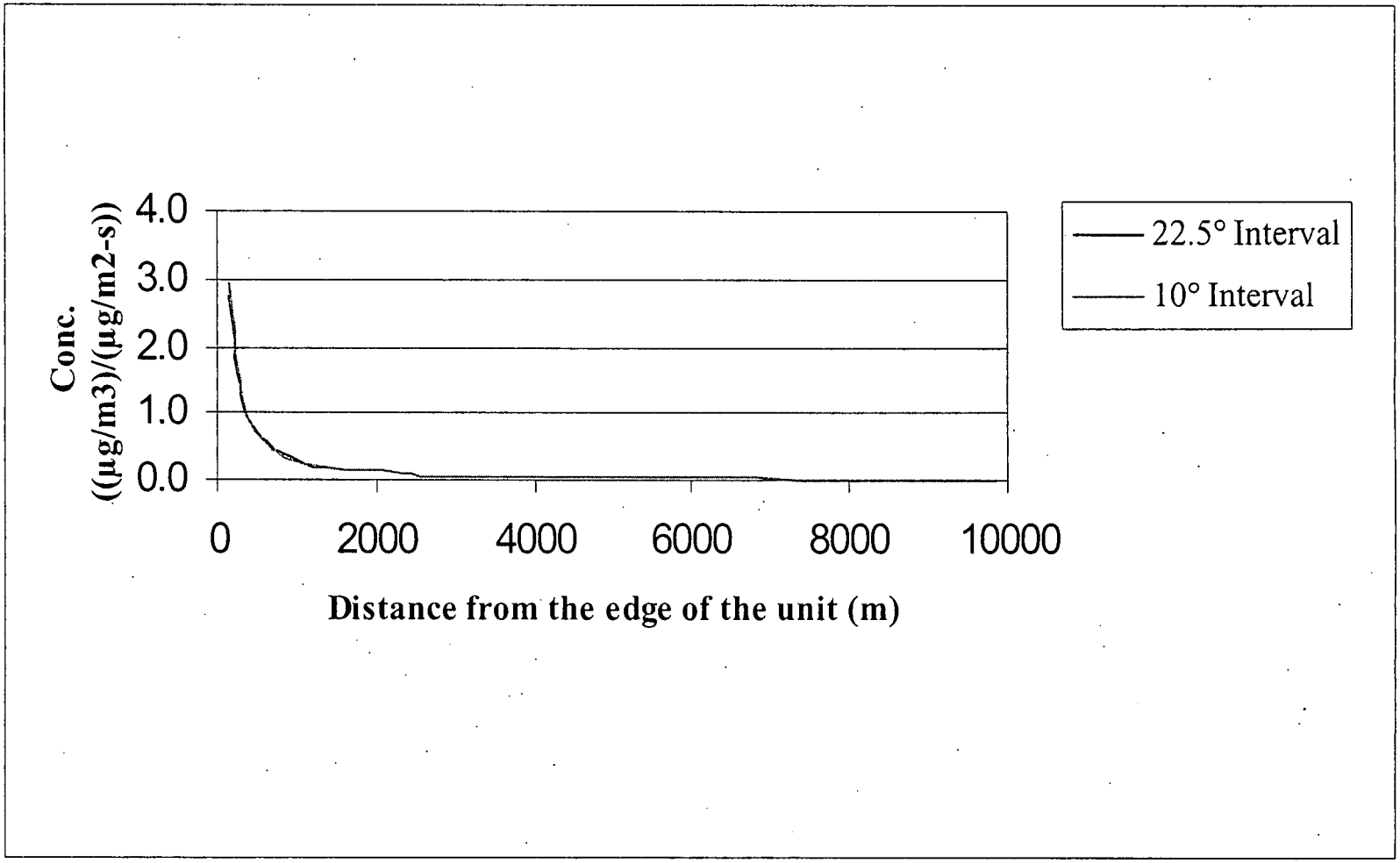


Figure C-8e. Maximum Concentrations (50th Percentile, LAU, Little Rock, AR)

122

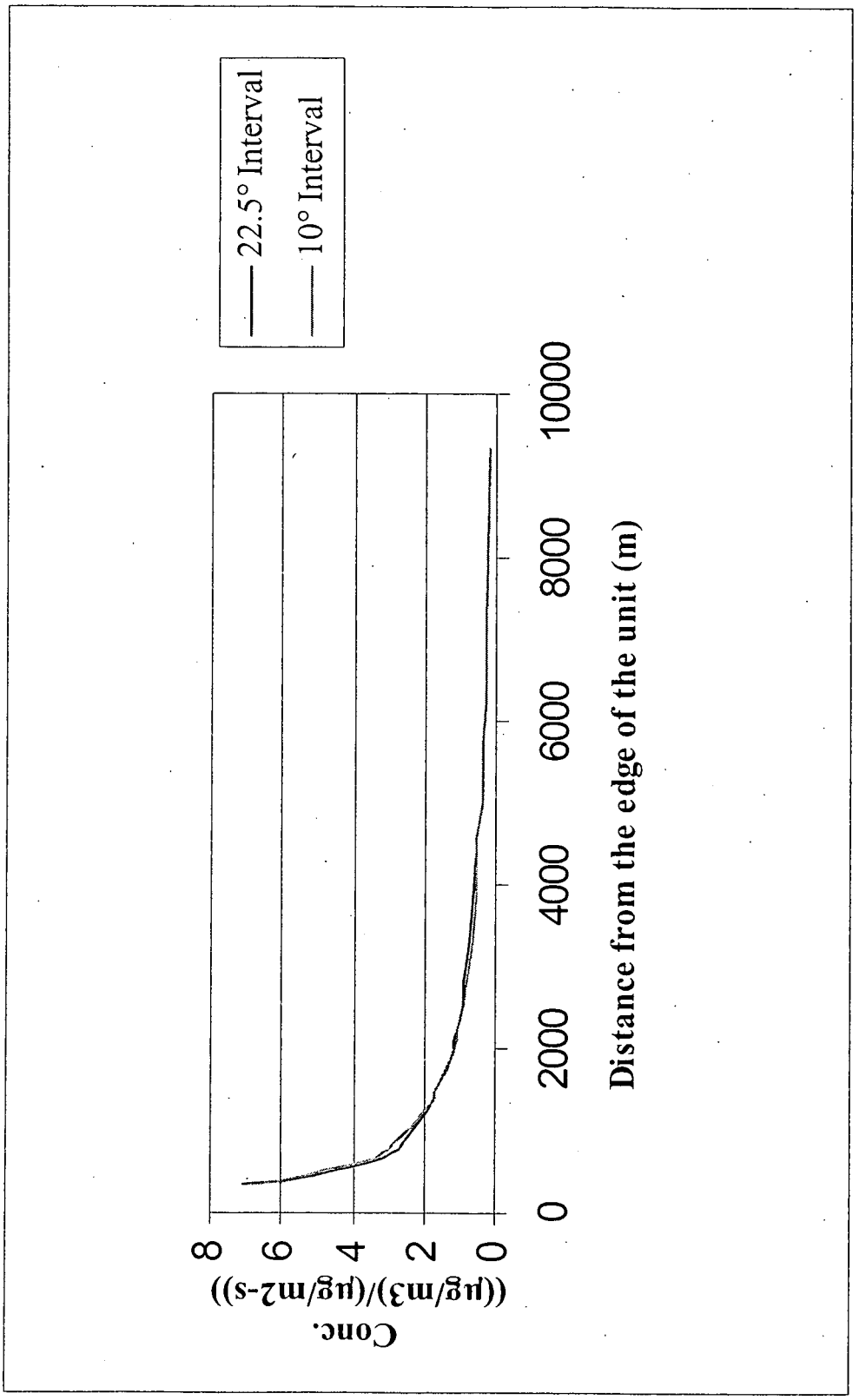


Figure C-8f. Maximum Concentrations (95th Percentile, LAU, Little Rock, AR)

123

C.4 An Analysis on Windroses at the 29 Sites

The hourly meteorological data from the 29 meteorological stations used in the Air Characteristic Study were used to generate windroses. A windrose consists of 16 directions, with the angle between any two adjacent directions being 22.5°. The prevailing wind directions for the 29 meteorological stations were counted to estimate the number of entries in each wind directions category. The results are presented in Figure C-9.

The narrowness of the most frequent wind directions for each of the 29 meteorological stations was examined. Based on the narrowness (or angles) of the most frequent wind directions, four categories were used to distinguish the windroses for the 29 meteorological stations. The four categories of windroses are:

- Narrowly distributed: most frequent wind directions no greater than 45°
- Moderately distributed: most frequent wind directions no greater than 90°
- Evenly distributed: no obvious predominant wind directions
- Bimodally distributed: most frequent wind directions are from two opposite directions.

The number of meteorological stations in each category is given in Table C-3. Figure C-10 gives some examples of windroses for each category. The windroses for the 29 meteorological stations are available and can be provided upon request.

An examination of the windroses and the maximum unitized annual average air concentrations from the Air Characteristic Study revealed that the sites with high concentrations are those with narrowly distributed wind directions. Simply put, persistent wind direction consistently blows pollutants from the source to the same receptors. Therefore, the more often the wind blows in a certain direction, the more likely high cumulative concentrations will occur at sites in that direction.

Air concentrations from a source are inversely proportion to windspeed. Given the same distribution of wind directions, a site with lower windspeed will have higher concentrations. The windroses show that, in the prevailing wind direction, the percentage of light wind occurring at a site with narrowly distributed wind directions is often higher than that at a site with evenly distributed wind directions. Therefore, we can conclude that a site with narrowly distributed wind directions will most likely produce the highest long-term average air concentrations.

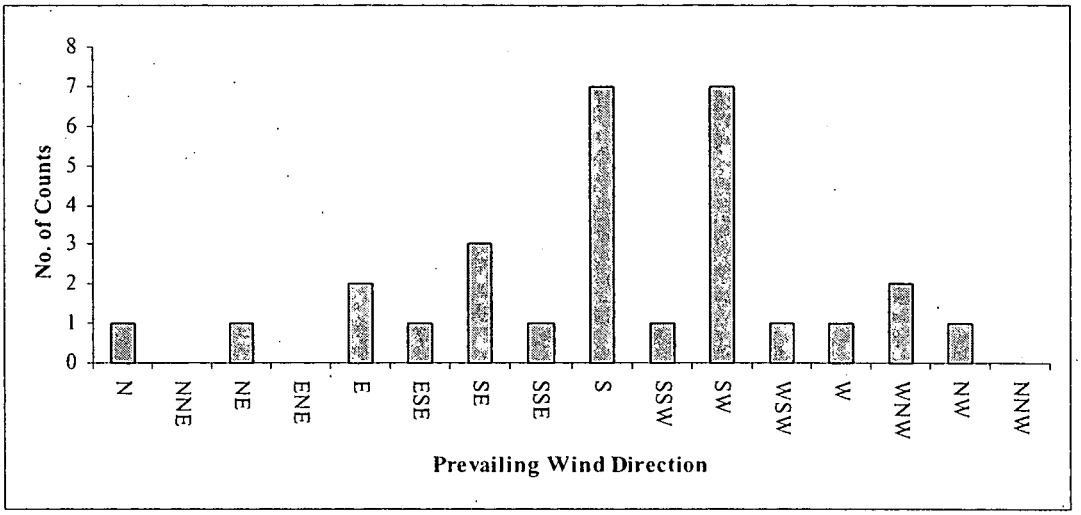
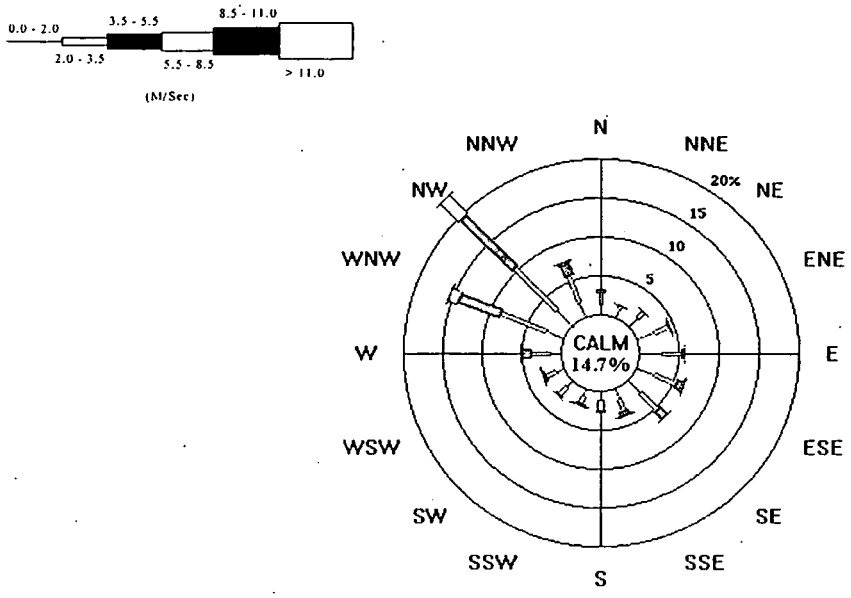


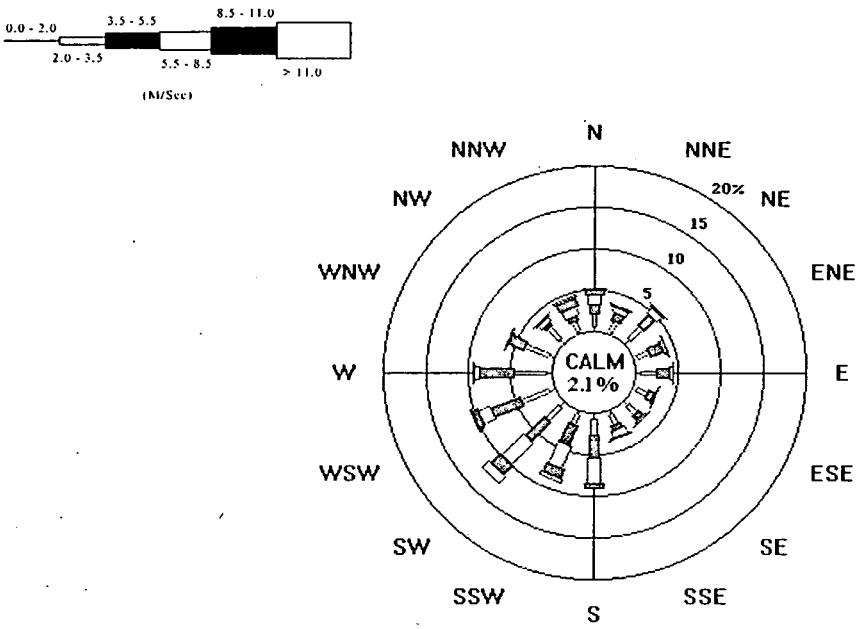
Figure C-9. Counts of Prevailing Wind Directions in Each Direction

Table C-3. No. of Met Stations with Different Shapes of Windroses

Shape of Windrose	No. of Stations
Narrowly distributed	10
Moderately distributed	4
Evenly distributed	6
Bi-modally distributed	9



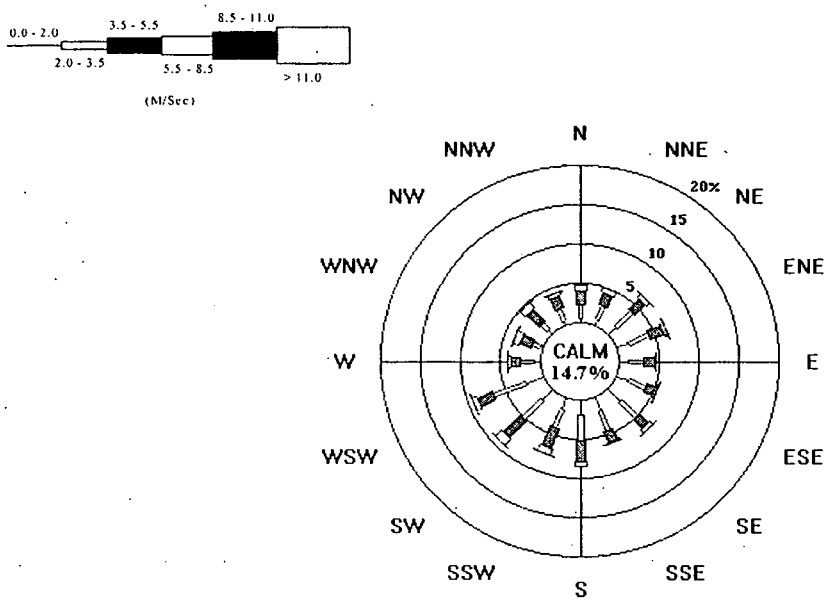
Narrowly Distributed Windrose



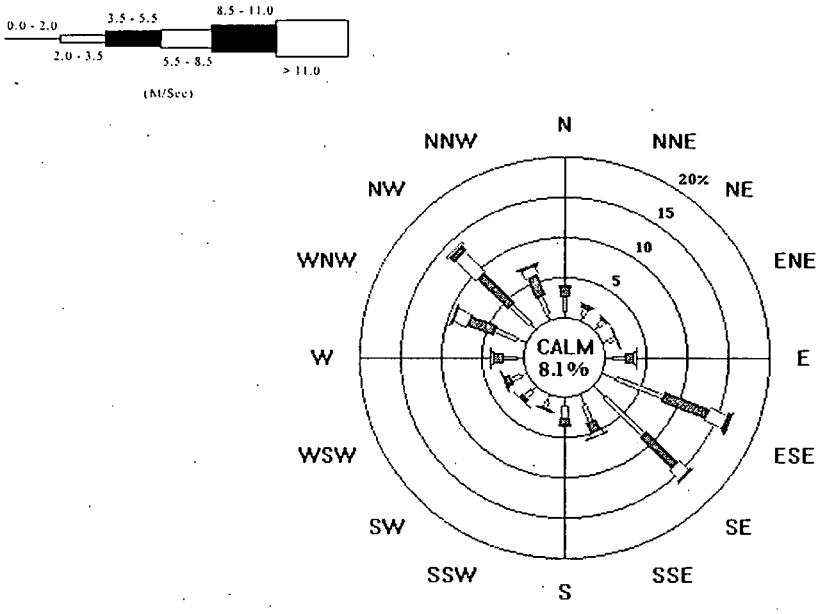
Moderately Distributed Windroses

Figure C-10. Examples of Different Shapes of Windroses

126



Evenly Distributed Windrose



Bi-modally Distributed Windrose

Figure C-10 (Continued). Examples of different Shapes of Windroses

127 | 127